DEVELOPING ECOLOGICAL GUIDELINES FOR MANAGEMENT AND RESTORATION OF WESTERN JUNIPER WOODLANDS[1]

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SUMMARY

It is suggested that the development of ecological guidelines for management and restoration of western juniper woodlands begins with clear comprehensive statements, preferably written, of desired future conditions (goals and objectives), followed by statements delineating the ways and degrees in which the present situation is lacking (problem statement). The next step suggested is an analysis of the resources available on the site including trees, understory plants and soils. This is followed by an analysis of how the site presently functions, with particular emphasis on aspects of the site that control key processes such as erosion and infiltration as well as plant reproduction, establishment, growth and succession. Finally, it is suggested that an analysis of site linkage to the surrounding landscape be made so that there is sense of the degree to which, the site influences or depends upon the surrounding landscape. From these statements and analyses actions can be designed that will facilitate those processes needed to reach desired future conditions.

INTRODUCTION

Attempts to develop site specific ecological guidelines for the management and restoration of western juniper woodlands are always underlain by a variety of assumptions. These assumptions may be derived from expectations, desires, and objectives that are unrealistic in terms of either the supporting science or the available resources. Assumptions are likely based on social values, understanding of ecological theories, and interpretation of ecological data, each of which need to be kept as distinct as possible (Scarnecchia 1995, Tausch 1996). We do not have comprehensive, whole system, ecological research data available on our arid lands. Therefore the land manager is faced with the formidable task of formulating suitable assumptions, developing reasonable objectives, and applying guidelines in an appropriate manner from shifting societal values, not so succinct and frequently conflicting ecological theories, less than adequate research data, as well as from observations and experience.

However, ecological principles and guidelines applicable to the management and restoration of woodlands can be found in, or can be derived from, a variety of comprehensive publications on range management, grazing management, rangeland improvement, and game range improvement. More specifically, Evans (1988) considered many ecological relations in

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[1] For a full length version of this paper see: Eddleman, L.E. 1999. Ecological guidelines for management and restoration of pinyon and juniper woodlands. In: Monsen, Stephen, B; Stevens, Richard; Tausch, Robin, J; Miller, Rick; Goodrich, Sherel. 1998. Proceedings: ecology and management of pinyon-juniper communities within the interior west. 1997, Sept. 15-18, Provo, UT. Gen. Tech. Rep. INT-GTR-000. Ogden, UT: U. S. Department of Agriculture, Forest Service, Intermountain Research Station.

developing strategies for management of pinyon-juniper woodlands for a variety of products and values.

Ecological guidelines must contain both mid-scale site qualities and broad-scale landscape qualities. When guidelines are developed for an ecological site (U. S. Department of Agriculture 1997) mid-scale, it is assumed that this site is unique within itself , however in addition it should be considered equally unique in its connectivity to surrounding sites within a landscape, broad-scale. Connectivity has been defined as the "degree to which patches of a given type are joined by corridors into a lattice of nodes and links" (Wiens et al. 1993). In woodlands connectivity could be considered as the degree to which a site is functional linked to other sites or landscape units of the system. At a very basic level each site functions either as a source of materials, functions to transport materials, or it functions in storage of landscape materials. The linkage is both spatial and temporal and operates through input, transfer, and output mechanisms of resource materials such as water, sediment, nutrients, energy, flora and fauna, etc. (Schlesinger et al. 1990, Tongway 1990, Wilcox and Breshears 1995).

Establishment of guidelines for management and restoration of western juniper woodlands implies that standards exist and that we are capable of directing or redirecting ecological processes to attain those standards. Nevertheless, setting ecological standards is a distinctly human process and is therefore an arbitrary value judgment that shifts with changing personal and societal values. We would like to think that western juniper woodlands have their own inherent ecological standards, and at certain temporal-spatial scales perhaps they do. Words such as "natural" and "healthy" pervade the literature, and our concept of these words, as applied to a particular area, provides the driving force for the establishment of arbitrary, not scientific, standards. However, Lawton (1997) has pointed out that "ecosystems change continuously at all time-scales". Standards and guidelines developed around concepts of sustainability should be called into question and carefully evaluated, particularly as to their spatial-temporal scale. Recognition of change and the processes associated with change should allow us to better derive our area goals, objectives, and standards more realistically focusing on trajectories and functional standards (Lawton 1997, Miller and Wigand 1994, Tausch 1996,).

When we select key functional attributes they should be the internal attributes of the ecosystem that control rate and magnitude of those processes that can lead to degradation, or vice-versa to desired conditions on the site in question. Functional attributes may be difficult to measure but several surface structural components have been found to be useful indicators of site function (de Soyza and others 1997). Approaches suggested by the Committee on Rangeland Classification (National Research Council 1994) are a good step toward the application of functional ecology to determination and management of rangeland health and are applicable to western juniper woodlands.

Development of ecological guidelines is at least a four-step process. The first step is the development of a set of goals and objectives. Secondly, a set of problem statements needs to be developed. This should be followed by a structural and functional analysis of the land area under consideration for treatment. Finally, a functional analysis relating the area under consideration to the landscape as a whole is needed. These are not new concepts but serve rather as a reminder since logical systematic approaches are needed to avoid pitfalls.

GOALS AND OBJECTIVES

(what the desired future conditions are)

Clear, concise goals and broad objectives are required at the outset of any management or restoration action. Establishment of goals and objectives requires that the desired pattern and process have been determined at both the site level and landscape-level for the land area under consideration for management change or restorative action. Examples of questions that need to be asked and answered include those below:

- What are the desired short-term and long-term ecological goals?
- What are the objectives and what is the spatial-temporal framework for measurement? (Literally, how many units per unit of space per unit of time must be specified)
- How do the objectives for the land area under consideration fit within the goals for the landscape as a whole?
- What are the assumptions that underlie each objective? (Why have I selected this objective and what gives credence to it?)

Unless considerable thought is put into making sure these questions have been answered, conflicts and cross-purposes may not be identified and the reality of success may be illusive. Ecological goals for woodlands should include components of sustainability and productivity as well as components of conservation of natural abiotic and biotic resources. Goals and objectives must be definable and achievable and the latter must be measurable. They should provide a clear statement of the desired direction of change, if any, for soils, plants and animals, and identify desired future woodland conditions. The degree of success should be measurable on both temporal and spatial scales. Knowing the degree to which one's assumptions rely on social values, ecological theory, ecological data, and observation and experience will help adjust confidence in reaching the goals and objectives.

PROBLEM STATEMENT

(what is wrong)

The next step in the process is to state precisely and clearly the apparent ecological problem(s) as it exists on the land area under consideration. This statement should be as comprehensive as possible and based on field observation and as much site-specific and local information as is available. It should identify those structural components that control rates and amounts of change and those processes that appear associated with degradation or with barriers to the achievement of future desired conditions. Degradation as used in this paper is the degree to which pattern and process have been altered and the land area under consideration rendered less favorable for the desired plant community or communities identified in the goals and objectives (modified from Tongway 1990). Some examples of questions from which a problem statement can be developed follow.

- What is there too much of or too little of? Why?
- What is wrong with the pattern? Why?
- What direction are things going or what processes are dominating the target site? Why?
- What are the underlying assumptions of each statement? (If a statement is made that there is approximately two times more bare ground than there should be, what assumptions have been brought to the statement formulation? Some examples could be: bare ground has a negative connotation, local observations suggest it, experience from similar sites suggests it, concepts and ecological theories favor it, interpretation of research data and established guidelines support it.

SITE ANALYSIS

(the resource materials present to work with)

Kind of Woodland - An inventory and analysis of the temporal and spatial nature of the woodland on the land area under consideration is needed since the kind of woodland presently on the land area determines to a large degree the management and restoration practices to be applied. Examples of questions that need to be asked and answered include those below.

- What is the age class structure of the trees? Are woodlands old, new or developing?
- What is the size class-spatial structure of the trees? Are woodland stands open or closed?
- What degradation processes are apparent, if any?

Old woodlands have a variable number of very old trees present and the land area appears to have been in woodland for several hundreds of years. Pattern of plants, soils and animals, and processes, both biotic and abiotic, may appear spatially and temporally controlled by the tree component of the woodland when the stands are closed. The tree component may appear to be stable; however, in the last century degradation in these woodlands may be ongoing at several scales.

New woodlands are those that have largely developed this century and there is no indication they were present previously. These woodlands contain mature trees; the larger individuals may have recently begun to slow down their growth rates but mortality factors are not strongly in evidence. Patterns and processes in these communities may be largely under the control of the woodland tree component. Degradation processes, if present, are likely multi-factored and complex.

Developing woodlands should be identified. These are areas that are being newly invaded by trees, but have not as yet reached the stage where internal patterns and processes are controlled or driven by the tree component.

Understory Vegetation - The composition and density of understory vegetation components must be determined on each functional unit within the land area under consideration in order to predict rates and amounts of transition in pattern and process toward the desired state. Examples of questions that need to be asked and answered include those below.

- Which species and functional groups of understory plants are present and which ones are absent?
- Is the apparent vigor or health of the shrub species sufficient to maintain them in the community or has a threshold been crossed and extinction expected?
- Given the density of long-lived herbaceous and shrub species, how will they respond to disturbance factors, both natural and man caused?
- What are the assumptions made in answering each question?

Functional Attributes - The local land area under consideration may be made up of one to several ecological sites, or one to several distinct functional landscape units. The area of interest must be delineated to assess functional attributes. Example questions that need to be asked and answered at both the mid-scale and broad-scale include those below.

- What ecological sites and functional landscape units are present?
- What structural factors currently dominate and control on-site pattern and process, and how do they do so?
- What causal factors led to current site conditions?
- What is the degree of degradation?
- What ecological thresholds currently restrict transitions to goal oriented ecological conditions?
- What assumptions were made in answering each question?

Thinking in spatial-temporal scales will greatly aid in understanding how an area functions. Spatially scaled functional units suggested by Wilcox and Breshears (1995) or those suggested by Anderson and Hodgkinson (1997) are very useful. Temporally scaled functions should at least include key seasonal aspects of moisture input as well as plant growth, reproduction, and establishment.

Basic processes of concern are those associated with the hydrologic cycle, nutrient cycle, and energy flow. Also of primary concern are successional processes in the functional groups of plants and processes associated with functional structures for animal habitat (National Research Council 1994, U. S. Department of Agriculture 1997).

Identification of the causal factors of degradation to pattern and process on a particular land area is complex. Additionally, degradation of one area may be caused by treatments on other areas. Such a relationship is particularly evident in riparian systems (Briggs 1996), but is also true for most upland systems. The degree to which certain ecological structures function as thresholds restricting transition to desired future conditions must be assessed (Laycock 1991, Reitkerk and van de Koppel 1997, Reitkerk et al. 1997, Tausch et al. 1995, Westoby et al. 1989). The degree of tree dominance, present and potential dominance of invasive alien plant species, lack of plant species, and individuals of particular species to respond, shallow soils, clayey or sandy textured soils, slopes receiving direct solar radiation at key seasons, surface soil loss, high surface water runoff, and intense spring frost action constitute some of the thresholds needing evaluation in western juniper woodlands. Threshold assessment provides the basis for determining treatment type and treatment magnitude, management changes and additions or deletions of organism and abiotic materials.

LANDSCAPE LEVEL ANALYSIS
(how the area is connected to the landscape)

A complete spatially temporally scaled landscape-level analysis relative to the land area or target site in question is necessary to assure selection of the right treatment area and right treatment and to prevent negative reactions on and off-site. Example questions that need to be asked and answered include those below.

- How does the landscape function as a whole?
- What are the links (connectivity) from the target site to adjacent and removed sites and functional units in the landscape?
- As to the natural resources of the landscape, which areas are source-areas, which areas are transport areas, and which areas are storage areas that intercept or accumulate resource materials?
- How does the particular site or land area under consideration fit functionally into the landscape?
- What reasons are there to believe that the area under consideration should function differently than it does?
- What are the assumptions made to support the conclusions?

Consideration of management units and treatment areas in isolation is not acceptable. It may be the case for a particular area of land that initial floristics and relay floristics are mechanisms determining composition of plant species over particular time scales. However, each area is linked to other areas in terms of a variety of critical resources that may strongly influence plant densities, vigor and energy values as well as rates and magnitude of processes.

In landscape analysis, emphasis may need to be placed on water flow, paying particular attention to the physical and biological structures that control and dominate the water cycle. Of critical concern is the ecological function of each land management area, ecological site, and functional unit for which restoration treatments are proposed. The degree of movement and accumulation of critical resources is hypothesized to be the principle mechanism controlling threshold levels of response in arid and semi-arid systems (Anderson and Hodgkinson 1997, Burke et al. 1995, Reitkerk et al. 1997, Tongway 1990, Ludwig et al. 1998).

MANAGEMENT AND RESTORATION ACTIONS

Following the structural and functional assessment of ecological factors, scaled from the functional unit up to the landscape, the problem statement needs to be revisited and revised. Once this process is complete, site-specific action guidelines of what, when, where, and how can be addressed. Common actions considered in the management and restoration of western juniper woodlands, such as burning, mechanical removal, seeding, and grazing should draw their ecological guidelines from a complete set of goals, objectives, and land analysis.

For example, questions as to grazing by domestic livestock can be answered only by asking if grazing will function to meet goals and objectives for the target site given the present structural and functional conditions of the resources of that site and influential off-site areas. More specifically, consideration of fall grazing requires that the prescription meet short-term and long-term goals and objectives, and that appropriate structural and functional resources in time and space are available to do so. For instance, fall grazing may be acceptable when the focus is on water capture and storage during the growing season, however, when capture and storage of winter and early spring moisture inputs are considered, fall grazing may become unacceptable.

Consideration as to grazing rest time after additions of species by seeding should be guided by goals, objectives, and land analysis. Future conditions, desired direction, rate of change, and site function should play a major role in the determination of what, when, where, and how grazing is to be used.

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THE IMPACTS OF JUNIPER ENCROACHMENT ON UNDERSTORY COVER AND DIVERSITY[2]

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SUMMARY

Western juniper (*Juniperus occidentalis*) has been actively invading shrub steppe communities in the Pacific Northwest during the past 120 years. The majority of these stands are still in a state of transition from shrub steppe to juniper woodland. In addition to different stages of stand development, juniper expansion occurs in different plant communities occupying different soils and topographic positions. Despite this high degree of variability, juniper woodlands are frequently treated generically in resource inventories, management, and wildlife habitat assessments. The goal of this study was to evaluate the influence of juniper dominance on plant community dynamics across several plant communities commonly impacted by western juniper encroachment in southeastern Oregon and northeastern California. The increase in juniper dominance had little impact on low sagebrush and an inconsistent affect on bitterbrush. However, as juniper dominance increased to about 50% of maximum canopy cover, mountain big sagebrush declined by nearly 80% of its maximum potential. Aspen also significantly declined as juniper dominance increased. Herbaceous cover and species diversity declined, and bareground increased with increasing juniper dominance in the mountain big sagebrush/Thurber needlegrass community. However, herbaceous cover on the deeper soils characterized by Idaho fescue and Columbia needlegrass did not decrease with increasing juniper dominance.

INTRODUCTION

In the Intermountain West the conversion of shrub steppe communities to juniper woodlands has been an actively ongoing process during the past 120 years. Over ninety percent of the 8 million acres of western juniper woodlands have developed in the last 100 years. Before settlement, juniper was primarily confined to rocky ridges or surfaces with sparse vegetation. However, newly formed juniper woodlands now occupy more productive sites with deep well-drained soils (Burkhardt and Tisdale 1969, Miller and Rose 1995). The replacement of shrub steppe communities with juniper woodland has been largely attributed to reduced fire frequency (Burkhardt and Tisdale 1976, Miller and Rose 1999). Heavy livestock grazing between 1880 and 1930 removed fine fuels (herbaceous biomass) which, previously had carried the fires.

Western juniper woodlands influence a large land area and occupy a broad array of environments. Despite the heterogeneity of the landscape occupied by juniper and the various stages of woodland development occupying these landscapes, juniper woodlands are frequently treated generically in resource management, inventories, and wildlife

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[2] This paper is a shortened version of a technical paper titled "Western Juniper Succession into Shrub Steppe: impacts on community dynamics". Submitted to the Journal of Range Management. Status:(in peer review).

habitat assessments. In addition, the classification of plant communities have frequently ignored woodland succession. Given the diverse landscapes occupied by western juniper, it is very likely that stand development and understory/overstory relations will be strongly influenced by soils, aspect, elevation, etc. The stage of woodland development will also affect composition and structure of the understory.

The objective of this study was to evaluate the influence of juniper dominance on plant community dynamics across several major communities common throughout the range of western juniper. Our specific objectives were to 1) evaluate the impacts of juniper on understory structure and composition across different plant communities common in the sagebrush steppe and 2) measure tree density and cover at full woodland development among different plant communities.

METHODS

The study area was located in southeastern Oregon and northeastern California. A combination of basin and range and weathered mountains of volcanic origin characterize the topography. The climate is cool and semiarid, characteristic of the northern Intermountain Region. Mean annual precipitation in the juniper zone across the study area typically varies between 12 and 16 in. (Taylor 1993). Three rangeland cover types were studied: low sagebrush, mountain big sagebrush, and aspen (Table 1). Aspen stands evaluated were associated with shrub steppe communities in the Steens Mountain and the south Warner Mountains below 7,000 ft. Study sites spanned from aspen on deep loamy soils to low sagebrush on shallow heavy clay soils, thus providing a variety of soils, landforms, and topographic positions. Elevation of study plots varied from 4750 to 6900 ft. One hundred and eight stands were measured across the study area. To evaluate the impacts of western juniper on understory composition and structure, we selected sites where disturbance (grazing, roads, etc.) appeared to have had minimal affect on community dynamics. We also selected sites where fire had been absent at least during the last 50 years.

Plant communities measured are reported in Table 1. Stages of woodland development were categorized into one of four successional stages: early, mid, late, and closed, based on annual tree growth and stand structural characteristics (Table 2). Two tree growth characteristics were used: annual lateral and terminal leader growth on sapling and full size trees. Stand structure characteristics used were tree cover, tree height, and proportion of live to dead shrub canopy. The key characteristic for stand closure was limited sapling leader growth, typically less than ½ in. We attempted to locate plots within each community in the four different phases of woodland development.

Tree and shrub cover and density as well as herbaceous and bareground cover were measured. Diversity and richness of plant species were also evaluated. Aspect, slope, and elevation were recorded for each stand and soils were described. Rill and gullies were noted if present on a site because they provided indirect evidence of soil movement.

Table 1: Plant community types included in this study.

Community Type	Sample size	Elevation (ft) Range (mean)	General Aspect	% Slope	Soils A & B Horizons
Low sagebrush / Sandberg bluegrass	12	4860-5980 (5333)		<2	A: Shallow clay loam to clay B: clay
Mountain big sagebrush /					
Thurber needlegrass	15	5166-6525 (5698)	Southerly	8-22	A: Sandy clay loam to clay loam B: Sandy clay loam to silty clay loam
Bluebunch wheatgrass	5	5068-6199 (5625)	South to West	8-26	A: Sandy to clay loam B: Clay loam to clay
Idaho fescue	49	5000-6580 (5650)	Northwest to Southeast	0-45	A: Loam to clay loam B: Clay loam to clay
-Mountain snowberry/ Columbia needlegrass	12	5115-6900 (6050)	Northwest to Northeast	10-25	A&B: Loam
Quaking aspen	15	5840-6710 (6250)	North to East	8-35	A&B: Loam

Table 2. Juniper woodland successional stages in mountain big sagebrush sites.

Characteristics (Post Settlement Stands)	Early	Mid	Late	Closed
Tree Canopy	Open, actively expanding, cover ≤ 5%	Actively expanding, cover 6 to 20%	Canopy expansion greatly reduced, cover 21-30%	Canopy expansion stabilized, over > 30%
Leader Growth (Dominant Trees)	Good terminal & lateral growth	Good terminal & lateral growth	Good terminal growth, reduced lateral growth	Reduced terminal growth, lateral growth absent
Crown Lift[3] (Dominant Trees)	Absent	Absent	Lower limbs beginning to die where tree canopy > 40%	Present where tree canopy > 40%
Potential Berry production	Low	Moderate to high	Low to moderate	Low to near absent
Tree Recruitment	Active	Active	Reduced, limited primarily to beneath trees	Absent
Leader Growth (Understory Trees)	Good terminal & lateral growth	Good terminal & lateral growth	Reduced terminal & lateral growth; reduced ring growth	Absent, some mortality; reduced ring growth
Shrub Layer	Intact	Nearly intact to some mortality around dominant trees	≥ 40% dead	≥ 85% dead

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[3] Crown lift is the mortality of lower tree limbs usually due to shading from neighboring trees.

RESULTS

Juniper Cover and Density in Fully Developed Woodlands

Tree cover and density in fully developed juniper woodlands varied widely across different plant communities (Table 3). Woodland canopy cover of closed stands ranged from 19% in the low sagebrush/Sandberg bluegrass community to a high of 90% in aspen community. Tree density in closed juniper stands also varied widely across communities ranging between 26 and 700 trees/ac in closed stands. As woodland development approached stand closure, maximum density of young trees (less than 10 ft tall) declined (Fig. 1).

Table 3. Mean and range of juniper cover and density (trees > 10 ft tall) in closed stands for six community types. Means followed by different letters were significantly different ($p \leq 0.0001$) for cover or density between community types.

Community Type	% Cover (range)	Trees/Ha (range)
Low sagebrush/Sandberg bluegrass (n=4)	21^a (19-24)	34^a (26-45)
Mountain big sagebrush/ Thurber needlegrass (n=6)	34^b (25-41)	140^b (90-195)
Mountain big sagebrush/ bluebunch wheatgrass (n=2)[4]	43 (35-47)	140 (105-180)
Mountain big sagebrush/ Idaho fescue (n=15)	48^c (34-58)	194 (100-290)
Mountain big sagebrush-snowberry/Columbia needlegrass (n=1)[4]	63	360
Quaking aspen (n=3)	84^d (78-90)	535 (366-700)

Shrub and Aspen Canopy in Response to Juniper

Increasing juniper density had little affect on low sagebrush cover. This may partially be due to the open nature of these stands. However, there was a strong relationship between juniper and mountain big sagebrush canopy cover (Fig. 2). The shrub layer declined by nearly 80% of maximum potential in the mountain big sagebrush/Thurber needlegrass, mountain big sagebrush/Idaho fescue, and mountain big sagebrush-snowberry/Columbia needlegrass communities when juniper canopies reached 50% of maximum woodland cover. The response

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[4] Not included in analysis because n<3.

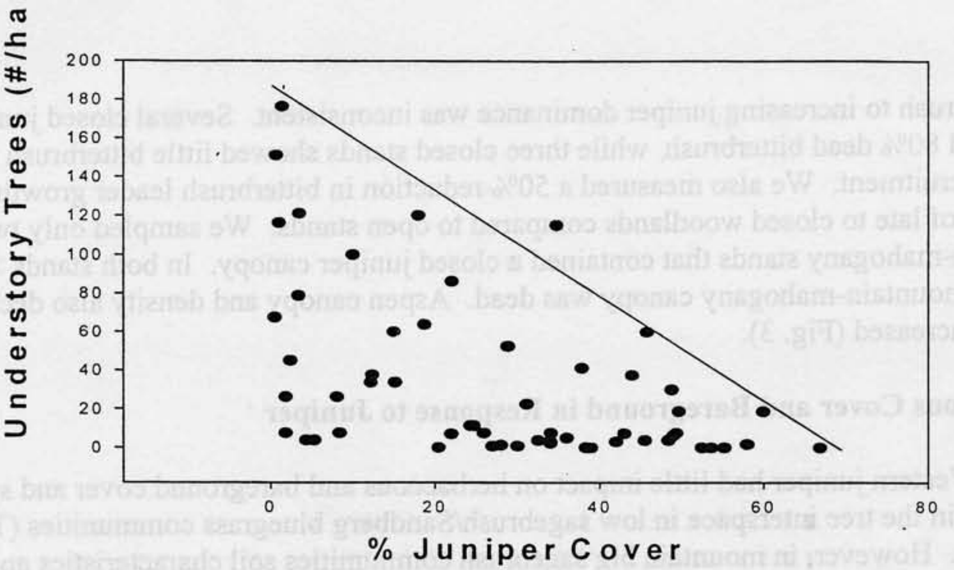

Figure 1. The relationship between understory juniper tree density (trees <3m tall) and mature overstory tree canopy cover for mountain big sagebrush communities. The line represents a boundary layer of maximum juvenile juniper trees occurring with varying juniper overstory canopy cover.

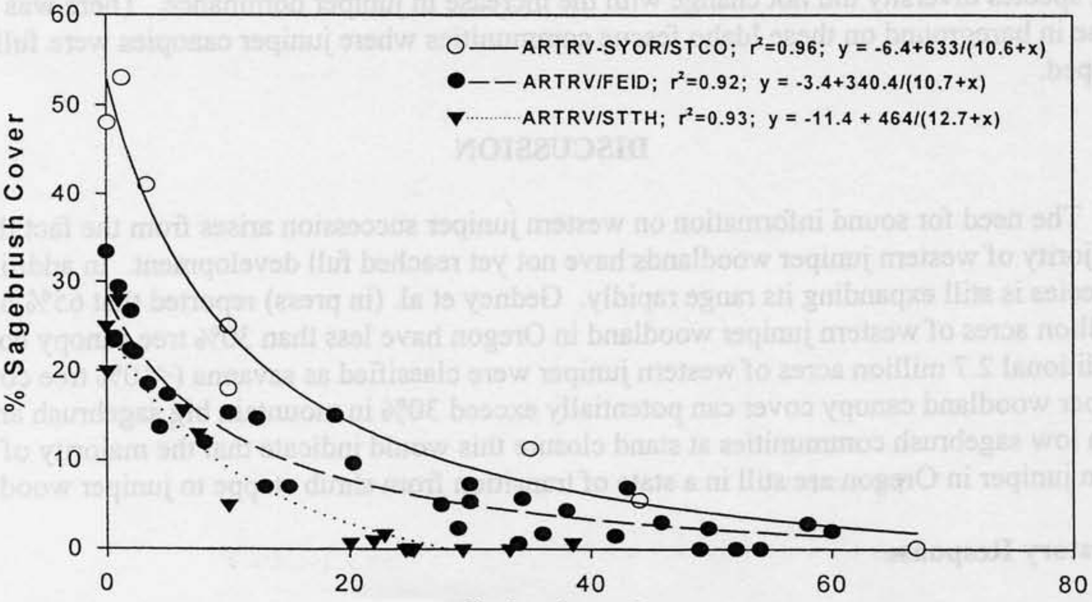

Figure 2. The relationship between juniper and mountain big sagebrush cover for the mountain big sagebrush/Thurber needlegrass (ARTRV/STTH), mountain big sagebrush/ Idaho fescue (ARTRV/FEID), and mountain big sagebrush-snow berry/Columbia needlegrass (ARTRV-SYOR/STCO) communities.

of bitterbrush to increasing juniper dominance was inconsistent. Several closed juniper stands contained 80% dead bitterbrush, while three closed stands showed little bitterbrush mortality and active recruitment. We also measured a 50% reduction in bitterbrush leader growth across the majority of late to closed woodlands compared to open stands. We sampled only two curlleaf mountain-mahogany stands that contained a closed juniper canopy. In both stands > 90% of the curlleaf mountain-mahogany canopy was dead. Aspen canopy and density also declined as juniper increased (Fig. 3).

Herbaceous Cover and Bareground in Response to Juniper

Western juniper had little impact on herbaceous and bareground cover and species diversity in the tree interspace in low sagebrush/Sandberg bluegrass communities (Table 4, 5, & 6; Fig. 4). However, in mountain big sagebrush communities soil characteristics appeared to determine the affects on juniper on understory vegetation. In the drier mountain big sagebrush communities characterized by shallow soils (often underlain by a hardpan between 12 and 24 inches below the surface) juniper significantly increased the amount of bareground (Table 5) as well as decreased plant herbaceous cover (Table 4 & Figure 5) and species diversity (Table 6). The dominant grass on these sites was often Thurber needlegrass. On deeper well drained soils characterized by an understory of Idaho fescue, increasing juniper had little affect on herbaceous cover. However, the affects juniper had on plant diversity varied across stands measured in the mountain big sagebrush/Idaho fescue community. In some stands a decline in species diversity as juniper increased was primarily due to a decline in the abundance of forbs. However, in other stands, species diversity did not change with the increase in juniper dominance. There was no increase in bareground on these Idaho fescue communities where juniper canopies were fully developed.

DISCUSSION

The need for sound information on western juniper succession arises from the fact that the majority of western juniper woodlands have not yet reached full development. In addition, this species is still expanding its range rapidly. Gedney et al. (in press) reported that 65% of the 2.2 million acres of western juniper woodland in Oregon have less than 30% tree canopy cover. An additional 2.7 million acres of western juniper were classified as savanna (<10% tree cover). If juniper woodland canopy cover can potentially exceed 30% in mountain big sagebrush and 20% in low sagebrush communities at stand closure this would indicate that the majority of western juniper in Oregon are still in a state of transition from shrub steppe to juniper woodland.

Understory Response

During woodland succession, the decline in mountain big sagebrush canopy is not proportional to the increase in juniper canopy. As juniper approaches 50% of maximum potential canopy cover in mountain big sagebrush communities, mountain big sagebrush

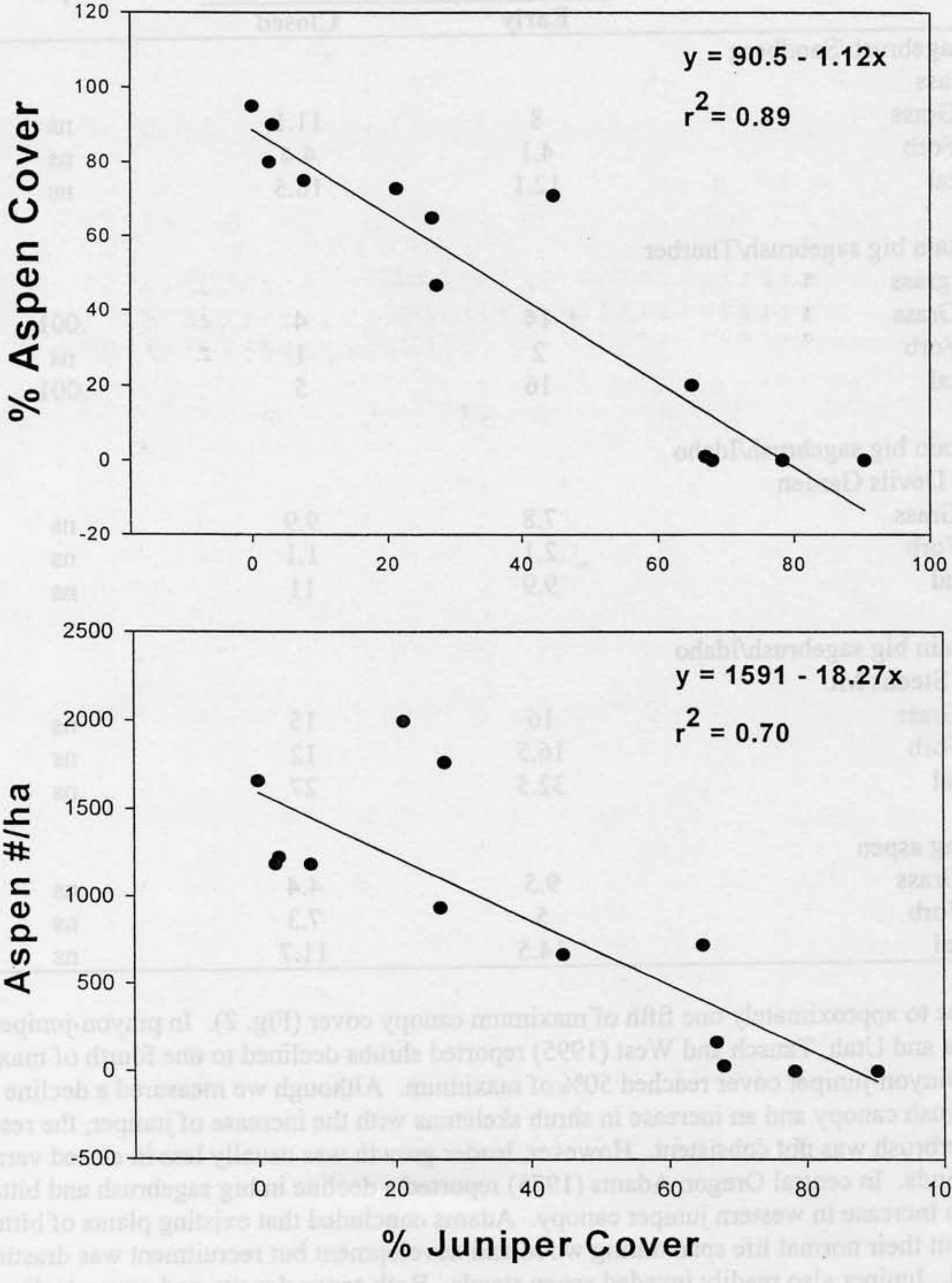

Figure 3. The relationship between juniper canopy cover and aspen canopy cover ($p \leq 0.0001$) and density ($p \leq 0.003$)

Table 4. Perennial grass and forb cover (%) for early and closed stages of woodland development across different plant communities (ns = not significantly different).

Community Type	**% Cover**		***p***
	Early	**Closed**	
Low sagebrush/Sandberg bluegrass			
P. Grass	8	11.1	ns
P. Forb	4.1	4.4	ns
Total	12.1	16.5	ns
Mountain big sagebrush/Thurber needlegrass			
P. Grass	14	4	.001
P. Forb	2	1	ns
Total	16	5	.001
Mountain big sagebrush/Idaho fescue Devils Garden			
P. Grass	7.8	9.9	ns
P. Forb	2.1	1.1	ns
Total	9.9	11	ns
Mountain big sagebrush/Idaho fescue Steens Mt.			
P. Grass	16	15	ns
P. Forb	16.5	12	ns
Total	32.5	27	ns
Quaking aspen			
P. Grass	9.5	4.4	ns
P. Forb	5	7.3	ns
Total	14.5	11.7	ns

declines to approximately one fifth of maximum canopy cover (Fig. 2). In pinyon-juniper in Nevada and Utah, Tausch and West (1995) reported shrubs declined to one fourth of maximum when pinyon-juniper cover reached 50% of maximum. Although we measured a decline in bitterbrush canopy and an increase in shrub skeletons with the increase of juniper, the response of bitterbrush was not consistent. However, leader growth was usually less in closed versus open woodlands. In central Oregon Adams (1975) reported a decline in big sagebrush and bitterbrush with an increase in western juniper canopy. Adams concluded that existing plants of bitterbrush lived out their normal life span during woodland development but recruitment was drastically limited. Juniper also readily invaded aspen stands. Both aspen density and cover declined as juniper canopy cover increased. In the absence of fire, juniper will likely continue to invade and replace aspen stands. Grazing by both domestic and wild large herbivores is also a major factor influencing aspen recruitment.

Table 5. Percent bareground in the tree interspace for five community types.

Community Type	**% Cover**				
	Open	**n**	**Closed**	**n**	**Prob**
Low sagebrush/Sandberg bluegrass	56	5	54	4	ns
Mountain big sagebrush/ Thurber needlegrass	55	5	90	7	.001
Mountain big sagebrush/ Idaho fescue (Devils Garden)	34	6	32.4	9	ns
Mountain big sagebrush/ Idaho fescue (Steens)	16	9	18	6	ns
Quaking aspen	5	8	3.8	3	ns

Table 6. Mean plant diversity indices within community types for early and closed juniper woodlands. Hills diversity numbers: N0 = species number; N1 = Shannon's index, the number of abundant species; N2 = Simpson's index, the number of very abundant species. Neither mountain big sagebrush nor juniper cover were included in diversity indices.

Community Type	**N0**	**N1**	**N2**
Low sagebrush/Sandberg bluegrass			
open (n=5)	35	8.4	5.9
closed (n=4)	37	9.5	6.4
Mountain big sagebrush/			
Thurber needlegrass			
open (n=5)	45	10.6	7.2
closed (n=6)	39	2.5	1.6
Idaho fescue (Devils Garden)			
open (n=6)	33	9.2	6.3
closed (n=9)	38	4.0	2.4
Idaho fescue (Steens Mt)			
open (n=9)	43	10.7	6.2
closed (n=6)	41	10.3	6.6
Quaking aspen			
open (9)	35	8.8	5.7
closed (4)	35	8.9	5.7

The response of herbaceous species to increasing juniper dominance did not always fit the common belief that as western juniper increases the herb layer declines. Across southeastern Oregon and northeastern California the response of herbaceous plant cover in mountain big sagebrush/Thurber needlegrass communities consistently declined as western juniper increased. The majority of these soils had a restrictive layer between 12 and 20 in. deep. We observed a very high density of juniper roots in the soil layer above the hardpan in our soil pits. In this same community, Bates et al. (1999) reported a 3 fold increase in herbaceous cover during the second growing season following western juniper removal. However, in the mountain big sagebrush/Idaho fescue community there was no significant difference in herbaceous cover between early and closed woodlands. These sites occupied deeper soils underlain by fractured bedrock. In the soil pits we observed a lower concentration of juniper roots in the upper 20 inches.

The amount of bareground in the tree interspace was greater in closed stands compared to earlier stages of development only in the mountain big sagebrush/Thurber needlegrass community. Although juniper cover averaged 34% in closed stands, little plant or litter cover protected the interspace, which accounted for 66% of the total area. We observed considerably more surface soil movement on this community than any other community.

However, the relationship of juniper cover and understory changes under poor grazing practices, where perennial herbaceous cover will significantly decline in juniper woodlands. We have observed significant increases in bareground across all juniper woodlands where past grazing practices have been abusive.

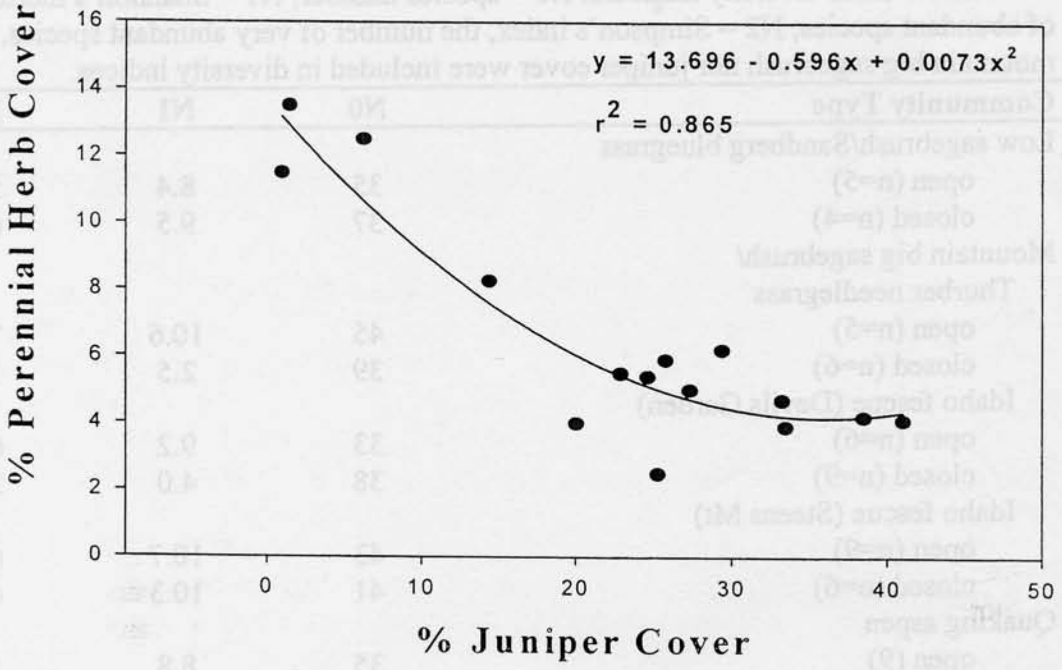

Figure 4. The relationship between perennial herb and juniper cover in the mountain big sagebrush Thurber needlegrass community.

Floristics and Diversity

We encountered 335 vascular plant species across the 105 stands we measured. The contribution of perennial forbs was highly variable among and within communities. Mean forb cover ranged between 1 and 16.5% among communities. On Steens Mountain, forbs accounted for a significant portion of the understory vegetation, while contributing ≤2% of the ground cover in northern California.

MANAGEMENT IMPLICATIONS

The identification of woodland successional stage and site variability in western juniper woodlands is extremely important when evaluating potential resource problems, assessing wildlife habitat values, and developing management plans. As shrub steppe communities are converted to juniper woodlands, community structure, composition, function, processes, and wildlife habitat suitability are altered. The conceptual model presented in Figure 5 (derived from Archer 1989) illustrates the conversion of a mountain big sagebrush steppe community to juniper woodlands in the absence of fire. During the early phases of woodland development, transition is easily reversible with fire. Juniper establishment during the early seral stages of shrub steppe is limited by a low density of shrubs. By the mid- to late stages of transition, a threshold is crossed when the natural reversal to shrub steppe communities by fire is unlikely. As shrubs decline the probability of a fire event intense enough to kill large juniper trees rapidly decreases. Juniper establishment declines with woodland maturity because of a decrease in safe seedling establishment sites provided by the shrubs. On-site seed input also declines because of the loss of the fruit crop through increasing tree competition. As juniper woodlands close and mast crops (Miller and Rose 1995) and shrubs are lost, fauna dependent on berries or shrubs also decline.

As community structure changes during woodland development, management options, wildlife habitat suitability, and important ecological variables such as hydrologic, nutrient, and energy cycles also change. Crossing ecological thresholds not only results in a significant reduction in the role of fire, but depending on the site may result in the loss of native plant species and loss of soils. Proper identification of the community and soils will indicate the effects of stand closure on a given site. The state of woodland transition can be identified by structural characteristics described in Table 1. Tree canopy cover and density alone are only of value if the range within a community has been defined. To improve interpretation, the spatial and temporal heterogeneity of a site or landscape in question must be identified when conducting inventories, designing research, or in developing management plans and classification schemes.

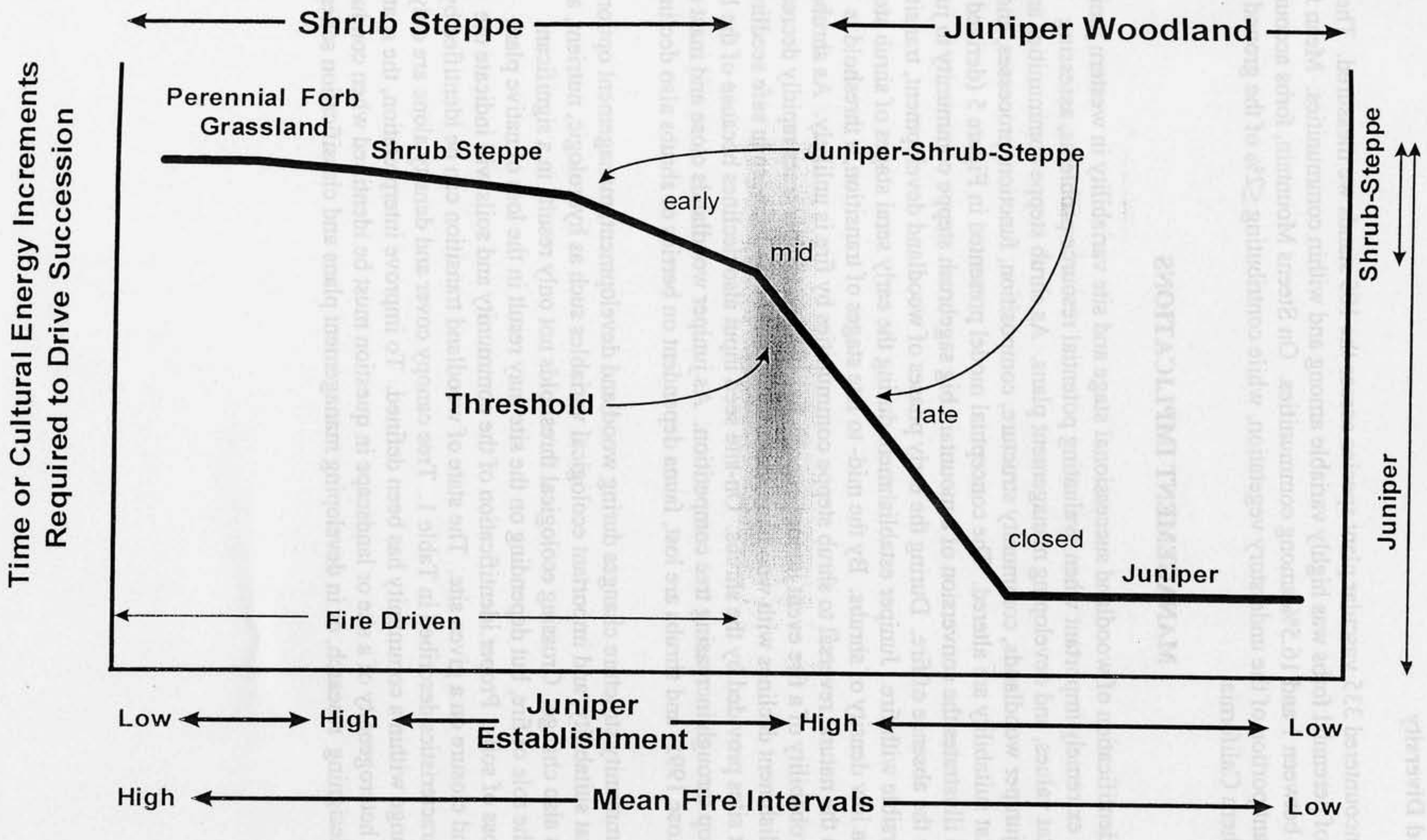

Figure 5. Conceptual diagram of changes in a shrub steppe community in the absence of fire (modeled after Archer 1989). In the absence of fire the abundance of shrubs decline as juniper trees gain dominance. A threshold has been crossed when understory fuels drop to a level where fire is unlikely to carry through the stand or generate enough heat to kill trees > 3m tall. The probability of the woodland crossing the threshold and reverting back to shrub steppe is very low in the absence of a major disturbance or very costly inputs. Tree establishment declines due to a decrease in seed input and safe sites provided by shrubs.

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JUNIPER AND WATERSHEDS

John C. Buckhouse

SUMMARY

Western juniper encroachment into shrub-steppe communities can have a significant effect on the water cycle. Sites where juniper encroachment fosters biomass concentration at the tree with increasing amounts of bare ground in the interspaces become drier from decreased infiltration of precipitation into the soil profile and increased surface flows, which quickly carry water off-site. Sites also become drier with increasing juniper dominance because of interception and evaporation, gully erosion, and a lowering of the capillary fringe associated with influent ground-water systems and desert streams. High transpiration rates can also occur on a site as an evergreen species encroaches onto a seasonal growth species' original habitat.

Role of Organics

In watershed management, the key phrase is "the soil's ability to capture, store and beneficially release water". This phrase has been bandied about since Hugh Barrett coined the phrase and I began to popularize it some 15 years ago. The phrase works because it reminds one of infiltration (capture), nutrient and water recycling (storage), and water quality in terms of regimen, erosion/sedimentation, bacterial relationships, nutrient fate, and water temperatures (beneficial release). But how does this relate specifically to juniper woodlands?

Watersheds may provide certain properties associated with water quality, but to do so it all begins with the "capture" part of this formula. Infiltration is defined as the movement of water across the air-soil interface, in other words, water moving from the atmosphere into the soil. This movement is influenced by a number of geologic, biotic, and abiotic factors.

Most textbooks suggest that these factors are soil texture, soil structure, slope, frost conditions, antecedent moisture relationships, soil colloids, and organisms. It is appropriate to note that while these biophysical parameters create the boundaries for the amount of water which can infiltrate at any given site, they all are ameliorated by the addition or removal of organics. Soil texture will not be changed in terms of its abiotic features given the addition of organics, but the distribution of those soil particles will be. Addition of organics will effectively increase pore space in any given soil textural class. Soil structure is defined by the development of "peds" within the soil profile. For all intents and purposes, organics are the "glue" which holds peds together and promotes their formation. Slope will be ameliorated at the microsite level by the retention/detention storage capacity of vegetation present on the surface, hence effectively holding precipitation in contact with the soil longer and enhancing its chance of infiltrating. Frost forms more quickly and freezes harder under conditions of bare ground, and as a consequence lowers infiltration rates. Vegetation is extremely valuable in absorbing incoming short-wave radiation and re-emitting long-wave radiation which slows or even prevents the formation of frost at the ground, leaving the soil more open to infiltration. Bone dry soils may actually repel raindrops when they finally do fall. If there is some residual soil moisture remaining, however, that will not be the case. Vegetation and incorporated soil organics are extremely valuable in conserving soil moisture and delaying evaporative soil moisture losses. Soil colloids are essentially the relationships of organic chemistry valence. Organics foster valence relationships which are conducive to open soils capable of increased infiltration. Finally, textbooks address the value of organisms in enhancing infiltration rates. These organisms are everything from earthworms to vegetation.

While it is clear that there are several biophysical features that limit infiltration, it is equally clear that under any given set of ecological conditions, standing vegetation, litter and duff at the soil surface, and incorporated organics within the soil profile ameliorate these limits and encourage infiltration as the organic component increases. It is also true that as organics are lost or unevenly distributed this enhancement of infiltration is lost.

Why Worry About Infiltration?

In a water budget, an arithmetic equation can be created:

$$P = I - SRO +/- S$$

Precipitation (P) equals infiltration (I) minus surface runoff (SRO) plus/minus any changes in surface storage (S).

To encourage subsurface storage of precipitation water and nutrient cycling, infiltration should be enhanced and surface runoff decreased. Decreasing surface runoff has the additional benefit of reducing the kinetic energy of the overland flows so that erosion and its subsequent decrease in soil production and fertility is minimized.

From a managerial point of view, anything that encourages the "capture" of precipitation (infiltration) is very positive. Not only will the water now be available for "storage" in the soil profile with its ultimate discharge as seeps, springs, transpiration through plants ("beneficial release") but also, the negative effects of rapid runoff with high kinetic energies and high potential for soil erosion, are reduced. Infiltration is the key to site productivity and function, and vegetation and its resulting organic component is the key to infiltration.

How Watersheds Relate to Juniper Sites

Western juniper (*Juniperus occidentalis*) is an extremely interesting and complex species: its adaptability and fluidity make it challenging and risky to stereotype. Several studies have been dedicated to understanding the ecology, form, function, and limits of this species. Since it has the ability to grow on a variety of soil types, under remarkable elevational and climatic extremes and aspect orientations - from scabland ridge tops to riparian bottoms - from alluvial fans to outwashes of snow cornices - in dry sagebrush sites well up into the coniferous zone- what is known about juniper may often seem disjunct and even contradictory. The species is, however, competitive over a wide variety of habitat types.

The growth patterns of juniper as it encroaches beyond its original, fire-restricting sites, demonstrates its competitive abilities. After evaluating the vegetation patterns with encroaching and maturing juniper, one notices that the on-site organic relationships change. The total biomass may not decrease, but the distribution of it does. Organics are now located in the tree itself. The remarkable and tenacious root system of the tree, coupled with its evergreen nature and ability to photosynthesize year-long, enable it to out-compete most native perennial grass, forb, shrub, and other tree species which have a definite dormant period. As the juniper tree out-competes its neighboring species and sequesters nutrients unto itself, the ground between trees becomes increasingly devoid of vegetation. This paucity of vegetation leaves those interspace sites vulnerable to splash impact of raindrops; to migration of fines to the surface, which in turn dry to hydrophobic "mud crusts"; to decreased infiltration into the soil profile; to increased surface runoff and, therefore, to higher kinetic energies capable of eroding the A and B soil horizons where most nutrient cycling takes place and where the most productive soil exists. Rill and gully erosion is common in these denuded interspace sites, causing further decline in site productivity to the extent that existing ecological states are overcome and ecological transitions to different, less productive, new states are created.

Oregon Plant Community Relationships

Infiltration work has been conducted over the years 1988-1992. Infiltration and sediment potentials have been recorded on ten Oregon ecosystems. Particularly the *Juniperus occidentalis* sites have consistently low infiltration rates, indicating high surface runoff flows, high kinetic energies, and therefore high erosion potentials (Buckhouse and Gaither 1982). In addition, encroached juniper sites with their low interspace vegetation cover have exponentially higher sediment/erosion potentials than do sites with greater ground cover resulting from more uniformly dispersed vegetation (Gaither and Buckhouse 1983).

Efforts by Eddleman, Miller, and other researchers in the last several years have refined much of the ecologic amplitudes of this species. The relationships between juniper and transpiration, evaporation, interception, soils, reproduction, plant community, and wildlife habitat has been studied and carefully documented. Yet the basic relationships of the watershed hold on sites where juniper encroachment fosters biomass concentration at the tree with increasing amounts of bare ground in the interspaces. These sites become drier from decreased infiltration of precipitation into the soil profile and increased surface flows which quickly carry water off-site. Sites also become drier with increasing juniper dominance from interception and evaporation, gully erosion, and a lowering of the capillary fringe associated with influent ground water systems and desert streams and high transpiration rates by an evergreen species encroached onto a seasonal growth species' original habitat. The removal of juniper trees on a mountain, big-sagebrush, Thurber-needlegrass site increased the length of the growing season for the herbaceous understory 6 weeks compared to adjacent uncut woodlands (Bates et al. 1999).

Solutions and Cautions

Since the fire regime which controlled juniper in presettlement times has been lengthened, it seems that the obvious solution to encroached juniper would be the reintroduction of fire. Some caution is advised. First, it may not be possible to ignite fires in mature stands where most, if not all, of the understory, fine fuels have been out-competed and lost. Next, even if a fire could carry, caution needs to be used on existing fuel loads. At the tree locations, where biomass is concentrated, hotter fires are probable. Compelling research exists that at soil surface temperatures between 300 and 600° F the colloidal relationships of the organic valence chemistry will change for the worst, creating hydrophobic soils which will not infiltrate at all. In Oregon, considerable effort has been expended to cut juniper trees then scatter limbs to create retention-detention microsites for watershed and herbaceous seed establishment, with an eye to reintroduction of fire somewhere down the line. Ideally, a wait of several years (perhaps as many as a dozen) should occur before reintroducing fire because she/he is well advised to have granted ample time for the herbaceous vegetation to establish prior to subjecting it to the disturbance of fire, ample time for nutrient cycling of the juniper needles and small twigs to occur, and time for the residual juniper seed source, lying in the soil, to have germinated and therefore be susceptible to the fire. All of these are watershed-based concepts with the goal of mimicking nature and the ground cover vegetation relationships which result in functional infiltration rates of the limited precipitation which falls on these semi-arid environments of the Great Basin.

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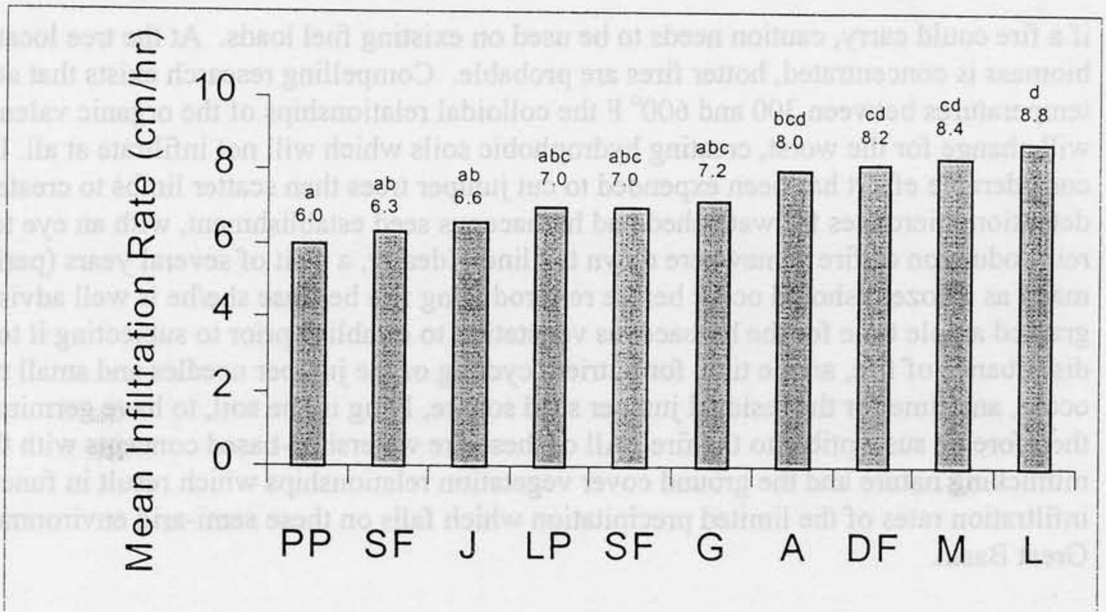

Figure 1. Mean infiltration rates of 10 ecosystems (different lower case letters indicate statistical difference P<0.10). Ecosystems: PP - ponderosa pine, SF - spruce-fir, J - juniper, LP - lodgepole pine, S - sagebrush, G - grassland, A - alpine, DF - Douglas fir, M - meadow, L - larch.

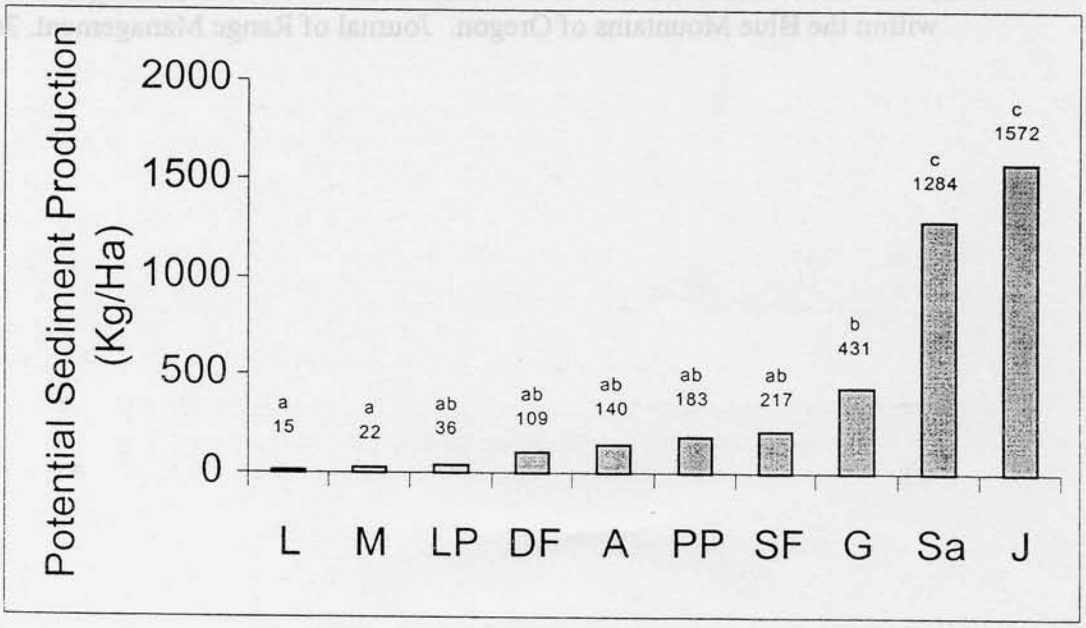

Figure 2. Potential sediment production in 10 Blue Mountain ecosystems. Different lower case letters indicate differences in statistical significance (P<0.10) Ecosystems: L - larch, M - meadow, LP - lodgepole pine, DF - Douglas fir, A - alpine, PP - ponderosa pine, SF - spruce-fir, G - grassland, Sa - sagebrush, J - juniper.

PLANT SUCCESSION IN CUT JUNIPER WOODLANDS: 1991-1998

Jon Bates, Rick Miller, and Tony Svejcar

SUMMARY

We monitored plant succession after tree cutting in grazed and ungrazed western juniper woodlands over an eight year period (1991-1998). Data from ungrazed woodlands span 1991-1997. Data for the grazed woodland in the late development stage were collected during 1991-1993, 1996, and 1998. The study sites were located on Steens Mountain in southeast Oregon. We selected eight 2 acre sized blocks of juniper woodland, and half of each block was cut in August 1991. Both grazed and ungrazed communities (4 blocks each) were rested from domestic livestock grazing in 1992 and 1993. Plots were grazed in the spring during the years 1994-1997 and were rested in 1998. Response trends for understory parameters were similar in grazed and ungrazed cut plots. Total basal cover, density, and biomass of the understory were greater in cut versus woodland treatments. Understory biomass was nearly 9 times greater in the ungrazed cut plots compared to ungrazed woodland in 1993, 1996, and 1997. In grazed plots, understory biomass was nearly 13 times greater in cut versus woodland plots in 1998. In all cut plots, annual grass cover, density, and biomass increased substantially by 1996 but was mainly confined to areas under cut trees (debris zones) and in litter layers around tree stumps. Perennial grasses, primarily squirreltail, preferentially established under juniper debris and around the old stumps. Understory response was not significant until the second year post-cutting (1993) when there were large increases in biomass, density, and cover in cut plots. Delays in understory response in the cut treatment were attributed to environmental conditions and seed production. Dry soil conditions in the spring of 1992 and 1994 contributed to the delay in understory response. The main increases in plant biomass, cover, and density in cut plots coincided with wet periods: the first increase occurring in 1993, and the second increase between 1995 and 1998. Because cutting is expensive, sites for treatment need to be carefully selected. Sites selected for cutting should contain adequate densities of understory perennial grasses. The density of perennial grasses necessary for recovery after cutting is dependent on site potential. A site lacking a perennial grass component and dominated by annual species will likely remain dominated by annual species unless seeded to perennials. The site used in this study probably requires rest or deferment during the growing season until perennial grasses have the opportunity to produce seed and establish seedlings. Grazing in late summer and fall appears permissible because plants are largely dormant during this time.

INTRODUCTION

Succession from sagebrush steppe to western juniper dominated communities has raised concerns over the effects of woodland development on plant community structure, composition, and diversity. Beginning in the late 1800's, western juniper woodlands have increased in density and spatial extent in central and eastern Oregon, northeastern California, and southwestern Idaho (Miller and Wigand 1994). Prior to Euro-American settlement, western juniper was largely confined to areas with shallow rocky soils underlain by fractured bedrock (Burkhardt and Tisdale 1969, Miller and Wigand 1994, Miller and Rose 1995). Recent juniper expansion has occurred on more productive soils occupied by mountain big sagebrush grasslands, riparian zones, and quaking aspen woodlands (Burkhardt and Tisdale 1969, Eddleman 1987, Miller and Rose 1995). Reduced fire frequency is cited as the main factor permitting the expansion of juniper (Burkhardt and Tisdale 1976, Evans and Young 1985, Miller and Wigand 1994). Succession to juniper-dominated communities is accompanied by reductions in understory productivity (West 1984, Vaitkus and Eddleman 1987), cover (Driscoll 1964), and diversity (Burkhardt and Tisdale 1969).

Many juniper dominated communities are relatively stable and resistant to all but the most severe fire conditions. Thus, natural or prescribed fire maybe eliminated as a management tool for restoring understory vegetation due to lack of fuels necessary to carry fires through juniper stands. Understory restoration in many juniper dominated communities is limited to mechanical treatments such as tree cutting.

Tree cutting to remove overstory competition is commonly used in areas occupied by western juniper. Cutting of juniper trees has resulted in greater understory biomass and cover (Vaitkus and Eddleman 1987, Rose and Eddleman 1994). Nonetheless, there is insufficient long term data documenting impacts of cutting in western juniper woodlands on plant community structure, composition, and diversity. Lack of long term ecological databases may hinder or misdirect decision making for understory restoration and commercial use of western juniper woodlands.

Understory dynamics in western juniper woodlands were assessed under grazed and ungrazed conditions after tree cutting. Cover, biomass, and density of understory species were monitored over a seven year period after cutting in 1991.

MATERIALS & METHODS

Study Site

The study site was on Steens Mountain in southeast Oregon. Elevation at the site 5000 ft and aspect is west facing with a 22% slope. Full occupancy of the site by juniper was indicated by the limited leader growth on juniper trees and low herbaceous cover. Mountain big sagebrush was largely eliminated from the site with only a scattering of old decadent shrubs remaining. On ungrazed plots juniper canopy cover averaged 25% and tree density averaged 119 trees ac^{-1}. On grazed plots juniper cover averaged 24% and tree density was 92 trees ac^{-1}. Bare ground was 97% in intercanopy zones and rill erosion was evident throughout the site.

The understory was dominated by Sandberg bluegrass, which constituted about 75% of total understory perennial plant cover. Other species characteristic of the site were bottlebrush squirreltail, bluebunch wheatgrass, Thurber needlegrass, basalt milkvetch, and pale alyssum. Perennial grasses on grazed plots were primarily of bluebunch wheatgrass and squirreltail. Perennial grasses on ungrazed plots were primarily of Thurber needlegrass and squirreltail.

Water year (October 1 - Sept. 30) precipitation at Malheur National Wildlife Refuge weather stations located 16 miles southwest at 4265 ft elevation and 18 miles northwest at 4100 ft elevation, have averaged 11 and 9.8 in. over the past 30 years. Soils on the site are 16 to 20 inches deep, rocky, and are clay loam in texture. Soils are underlain by a welded ash tuff of rhyolite/rhyodacite composition which blocks root penetration. Soils were classified as clayey-skeletal, smectitic, frigid, Lithic Argixerolls.

Experimental Design

Ungrazed Plots - The experimental design was a randomized complete block with 4 blocks (replicates) and 2 treatments, cut and uncut juniper woodland. Replicates were 0.8 ha (2 acres) in size and were selected for similarities in overstory/understory density and cover. Measurements of baseline vegetation characteristics were made prior to tree cutting in July 1991. Trees in half of each block were cut with chainsaws in August 1991. All cut juniper trees were left in place, a standard range improvement practice in eastern Oregon. Post-treatment measurements of understory characteristics began in April 1992. Livestock were excluded from the site from 1992 through 1998.

Grazed Plots - The experimental design was a randomized complete block with 4 blocks and 2 treatments, cut and uncut juniper woodland. Replicates were 0.8 ha in size. Pre-treatment vegetation characteristics were made prior to tree cutting. Trees in half of each block were cut in August 1991. Post-treatment measurements of understory characteristics began in April 1992. Livestock were excluded from the site in 1992, 1993, and 1998. Plots were grazed in the spring from 1994-1997 by 100-400 cow/calf pairs in the spring. Plots were in a pasture of about 200 acres and access was available to hay meadows below the site. The duration of use was approximately three weeks each year from late April to May 15.

Understory Sampling

Understory measurements were basal cover (perennial plants), canopy cover, density, and biomass. Understory plants were measured by species (except biomass) but are organized into five functional groups to simplify presentation of the results. The functional groups are Sandberg bluegrass, perennial bunchgrasses (e.g. Thurber needlegrass, bluebunch wheatgrass, squirreltail), perennial forbs, annual grasses, and annual forbs.

Biomass was sampled at 10 ft intervals using 10.75 ft^2 quadrates along two 150 ft transects in cut and uncut woodlands. Vegetation was clipped to a 1 in. stubble height. Understory plant density and canopy/ground cover was measured using 2 ft^2 quadrates. Basal cover of perennial plants was measured along five 100 ft line transects in 1991-93 and along four 150ft transects in 1994, 1997, and 1998. Transect length and number were changed in 1994 to conform with understory sampling protocols that are being used in other juniper woodland projects.

Sampling in the plots was conducted as follows:

Ungrazed Plots
    Biomass: 1992, 1993, 1996, 1997.
    Density: 1991-1997.
    Basal Cover: 1991-1994, 1997.
    Canopy Cover: 1993, 1994, 1997.

Grazed Plots
    Biomass:1992, 1993, 1996.
    Density: 1991-1993, 1998.
    Basal Cover: 1991-1993, 1998.
    Canopy Cover: 1993, 1998.

RESULTS

Understory Succession - Ungrazed Plots

Pre-cutting - Baseline measurements made before cutting in July 1991 indicated there were no differences in basal cover (Table 1) of herbaceous perennials or in understory plant density (Table 2) between plots that were left as woodlands and plots that were selected to be cut.

Post-cutting - *Sandberg bluegrass* - Cover (Table 1) and biomass (Table 3) of bluegrass increased between 1991 and 1993 in cut plots. Bluegrass cover and biomass were greater in cut versus woodlands during 1992 and 1993. By 1997, there were no differences between cut and woodland plots in cover, density (Table 2), and biomass. In cut plots cover and biomass of bluegrass declined dramatically between 1993 and 1997.

Table 1. Ungrazed Plots - Understory perennial plant basal cover (%) in cut and woodland treatments (1991-93 and 1997). Different letters denote significant differences for each species group in each year.

Species Group [1]	Treatment	1991	1992	1993	1997
Sandberg Bluegrass	Cut	1.4 ± 0.3 a	1.5 ± 0.4 a	2.8 ± 0.4 a	1.1 ± 0.2 a
	Woodland	1.2 ± 0.2 a	1.1 ± 0.2 a	1.2 ± 0.1 b	1.1 ± 0.1 a
Perennial Grasses[2]	Cut	0.9 ± 0.2 c	0.9 ± 0.2 d	2.6 ± 0.3 d	3.5 ± 0.3 d
	Woodland	0.8 ± 0.2 c	0.4 ± 0.1 c	0.4 ± 0.1 c	0.2 ± 0.1 c
Perennial Forbs	Cut	0.4 ± 0.1 g	0.4 ± 0.2 h	0.5 ± 0.2 h	0.3 ± 0.1 g
	Woodland	0.3 ± 0.1 g	0.1 ± 0.1 g	0.1 ± 0.1 g	0.1 ± 0.1 g
Total Cover	Cut	2.6 ± 0.4 y	2.8 ± 0.3 z	5.9 ± 0.3 z	4.8 ± 0.3 z
	Woodland	2.3 ± 0.3 y	1.7 ± 0.2 y	1.7 ± 0.3 y	1.5 ± 0.3 y

[1] Units are percent crown cover (%).

[2] Perennial grasses in this category are the taller tussock grasses, including bluebunch wheatgrass, Thurber' needlegrass, and squirreltail.

Perennial Bunchgrasses - In cut plots, the basal cover, density, and biomass all steadily increased between 1991 and 1997 (Tables 1-3). Perennial grass density increased by nearly 700%, basal cover increased by 350%, and biomass increased (1992 vs. 1997) by 2700%. In 1997, perennial grass biomass was 12 times greater in cut plots compared to woodlands. Basal cover and density of perennial grasses have been significantly greater in cut plots versus the woodlands in all years post-cutting. Perennial grasses, primarily squirreltail, preferentially established under juniper debris and around the old stumps (Bates et al 1998). In woodland plots perennial grass biomass increased between 1993 and 1997. However, basal cover of grasses has declined in woodlands over the same period.

Perennial Forbs - Perennial forb basal cover and density has not differed between treatments. However, in all years forb biomass was consistently greater in cut plots compared to woodlands (Table 3).

Annual Grasses - There was a distinct lag in annual grass (cheatgrass and Japanese brome) response to cutting. Densities of annual grasses did not differ in 1992 or 1993 between treatments. Annual grass density began increasing in 1994 and 1995 in cut plots. By 1996 and 1997 cover, density, and biomass of annual grasses increased exponentially in cut plots (Tables 1-3). The increase in annual grass has mainly been confined to areas under cut trees and around the old stumps (Bates et al. 1998).

Annual Forbs - Except for one year (1995) annual forb density has been significantly greater in woodland versus cut plots (Table 2). Annual forbs are larger in cut plots as evidenced by their greater biomass value in 1997 (Table 3). The lower annual forb density in cut plots results from lower densities of pale alyssum compared to woodlands (Bates et al. 1999).

Total ground cover - Ground cover was nearly twice as great in cut compared to woodland plots in 1993 and 1997. In 1997, ground cover in the cut treatment totaled 56% (juniper slash, 18%; duff and other litter, 20%; interspace herbaceous canopy cover, 18%) compared to 29% in the uncut treatment (juniper trees and litter, 24.5%; interspace herbaceous canopy cover, 4.5%).

Understory Succession - Grazed Plots

Pre-cutting - Baseline measurements in July, 1991 did not show any major differences plant cover (Table 4) or density (Table 5) between plots that were left as woodlands and plots that were selected to be cut. Perennial forb density and cover were slightly greater in woodland versus cut plots.

Post-cutting - *Sandberg bluegrass* - Cover (Table 4) and biomass (Table 6) of bluegrass increased between 1991 and 1993 in cut plots and were greater in cut versus woodlands.

Table 2. Ungrazed Plots - Understory density (1991-1997) in cut and woodland treatments. Different letters denote significant treatment differences for each species group in each year. In the cut treatment there were significant increases in perennial grass and annual grass densities particularly between 1994 and 1997.

Species Group[1]	Treatment	1991	1992	1993	1994	1995	1996	1997
Sandberg Bluegrass	Cut	8.0 ± 0.6 a	10.4 ± 1.1 a	11.1 ± 1.2 a	8.6 ± 0.7 a	5.9 ± 0.4 a	6.4 ± 0.2 a	9.8 ± 1.1 a
	Woodland	7.7 ± 0.7 a	10.4 ± 0.9 a	9.2 ± 0.9 a	13.3 ± 1.4 b	8.6 ± 0.5 b	11.5 ± 1.3 b	11.1 ± 0.6 a
Perennial Grasses	Cut	2.6 ± 0.3 d	2.2 ± 0.3 e	3.7 ± 0.4 e	5.1 ± 0.2 e	7.5 ± 0.9 e	10.0 ± 0.4 e	15.1 ± 1.4 e
	Woodland	2.1 ± 0.3 d	1.6 ± 0.2 d	1.9 ± 0.3 d	1.9 ± 0.4 d	2.2 ± 0.3 d	2.1 ± 0.5 d	1.7 ± 0.3 d
Perennial Forbs	Cut	0.4 ± 0.2 g	1.0 ± 0.4 g	1.7 ± 0.4 g	1.9 ± 0.3 g	2.9 ± 0.5 h	4.0 ± 1.5 h	4.5 ± 0.4 h
	Woodland	0.6 ± 0.3 g	0.6 ± 0.3 g	2.0 ± 0.3 g	1.2 ± 0.3 g	1.7 ± 0.4 g	1.9 ± 0.5 g	1.6 ± 0.3 g
Annual Grasses	Cut	0.5 ± 1.2 r	2.2 ± 1.1 r	3.7 ± 5.3 r	30.1 ± 3.7 s	77.0 ± 11.1 s	570.0 ± 61.0 s	442.0 ± 64.0 s
	Woodland	1.1 ± 3.4 r	2.5 ± 2.5 r	2.8 ± 2.7 r	3.4 ± 3.1 r	2.1 ± 1.2 r	5.9 ± 2.8 r	6.7 ± 2.7 r
Annual Forbs	Cut	12.6 ± 4.2 y	9.4 ± 2.3 y	40 ± 18 y	100 ± 18 y	139 ± 9 z	65 ± 6 y	207 ± 32 y
	Woodland	14.2 ± 1.4 y	83.5 ± 34 z	125 ± 43 z	174 ± 40 z	59 ± 14 y	102 ± 22 z	286 ± 37 z

[1] Units are number of plants per m^2 (or number of plants per 10.75 ft^2).

Table 3. Ungrazed Plots - Understory biomass (lb/ac) in cut and woodland treatments (1992-93 and 1996-97). Different letters denote significant differences for each species group in each year.

Species Group	Treatment	1992	1993	1996[2]	1997[2]
Sandberg Bluegrass	Cut	10.2 ± 1.0 a	165 ± 21 b	N.A.	36 ± 7 a
	Woodland	9.1 ± 1.6 a	41 ± 5 a		41 ± 5 a
Perennial Grasses[3]	Cut	16.6 ± 2.9 e	169 ± 25 e	N. A.	492 ± 65 e
	Woodland	7.1 ± 2.9 d	9 ± 3 d		41 ± 12 d
Perennial Forbs	Cut	13.8 ± 5.4 h	21 ± 6 h	N. A.	111 ± 10 h
	Woodland	0.4 ± 0.2 g	3 ± 1 g		28 ± 12 g
Annual Grasses	Cut	1.1 ± 0.5 j	5.0 ± 1.0 k	N. A.	211 ± 16 k
	Woodland	0.2 ± 0.2 j	0.5 ± 0.2 j		2 ± 1 j
Annual Forbs	Cut	N. M.	N. M.	N. A.	20 ± 2.6 s
	Woodland				0.7 ± 0.2 r
Total Biomass	Cut	41.7 ± 5.1 z	359 ± 35 y	931 ± 211 y	931 ± 85 y
	Woodland	16.6 ± 4.3 y	33 ± 5 z	91 ± 24 z	110 ± 29 z

[2] Biomass values in 1996 and 1997 include both current years production and standing dead material from previous years growth. Current years production in 1996 and 1997 is about 70% of the biomass values shown.

[3] Perennial grasses in the this category are the taller tussock grasses, including bluebunch wheatgrass, Thurber needlegrass, and squirreltail.

N. M. - not measured.

N. A. - not separated by species group.

Table 4. Grazed Plots - Understory perennial plant basal cover (%) in cut and woodland plots (1991-93 and 1997). Different letters denote significant treatment differences for each species group in each year.

Species Group[1]	Treatment	1991	1992	1993	1998
Sandberg Bluegrass	Cut	1.0 ± 0.1 a	0.9 ± 0.2 b	1.7 ± 0.3 b	0.7 ± 0.1 a
	Woodland	1.2 ± 0.1 a	0.8 ± 0.2 a	0.8 ± 0.2 a	0.5 ± 0.1 a
Perennial Grasses	Cut	0.4 ± 0.1 c	0.5 ± 0.1 d	1.6 ± 0.3 d	1.8 ± 0.1 d
	Woodland	0.6 ± 0.1 c	0.2 ± 0.1 c	0.2 ± 0.1 c	0.1 ± 0.0 c
Perennial Forbs	Cut	0.1 ± 0.0 g	0.3 ± 0.2 g	0.2 ± 0.0 g	0.1 ± 0.0 g
	Woodland	0.3 ± 0.1 h	0.2 ± 0.1 g	0.2 ± 0.1 g	0.1 ± 0.0 g
Total Cover	Cut	1.5 ± 0.1 y	1.8 ± 0.4 y	3.5 ± 0.3 z	2.7 ± 0.2 z
	Woodland	2.0 ± 0.2 y	1.2 ± 0.2 y	1.3 ± 0.1 y	0.7 ± 0.1 y

[1] Units are percent crown cover (%).

Table 5. Grazed Plots - Understory density (plants m^{-2}) in 1991-93 and 1998 in cut and woodland plots. Different letters denote significant treatment differences for each species group in each year.

Species Group[1]	Treatment	1991	1992	1993	1998
Sandberg Bluegrass	Cut	7.0 ± 0.8 a	7.8 ± 1.2 a	8.1 ± 1.0 a	7.1 ± 1.6 a
	Woodland	6.7 ± 1.4 a	7.0 ± 1.1 a	7.6 ± 1.4 a	6.9 ± 1.6 a
Perennial Grasses	Cut	3.5 ± 0.3 a	3.0 ± 0.3 b	3.0 ± 0.4 b	10.1 ± 0.2 b
	Woodland	2.9 ± 0.4 a	1.8 ± 0.2 a	1.6 ± 0.3 a	2.3 ± 0.4 a
Perennial Forbs	Cut	0.9 ± 0.1 a	2.4 ± 0.5 a	2.4 ± 0.4 a	5.2 ± 1.7 a
	Woodland	1.6 ± 0.4 b	2.5 ± 0.3 a	2.5 ± 0.5 a	3.1 ± 0.7 a
Annual Grasses	Cut	0.8 ± 0.3 a	2.0 ± 0.8 a	1.8 ± 0.6 a	547.0 ± 46.0 b
	Woodland	2.4 ± 1.9 a	10.8 ± 9.4 a	8.8 ± 7.1 a	10.1 ± 2.2 a
Annual Forbs	Cut	8.9 ± 1.3 a	97 ± 24 a	98 ± 24 a	266 ± 41 a
	Woodland	8.2 ± 1.5 a	293 ± 128 b	272 ± 81 b	527 ± 92 b

[1] Units are number of plants m^{-2} (or number of plants 10.8 ft^{-2}).

In 1998 there were no differences between cut and woodland plots in cover and density (Table 5) as bluegrass presence declined dramatically between 1993 and 1998 in cut plots.

Perennial bunchgrasses - In cut plots, basal cover (Table 4) and density (Table 5) increased between 1991 and 1998. In woodlands, perennial grass basal cover has declined and in all years was significantly less than cover in cut plots. Density increases in cut plots occurred during the grazing period. Perennial grass density has more than tripled since 1994. Woodland perennial grass density has not changed since 1991 but basal cover has declined by75%.

Perennial Forbs - Perennial forb basal cover and density were not different between treatments. In 1992 and 1993, forb biomass was greater in cut plots compared to woodlands (Table 6).

Annual Grasses - Annual grass response trend has been similar to ungrazed plots. Densities of annual grasses did not differ between treatments in 1992 and 1993. In 1998, annual grass density was 55 times greater in the cut versus woodland treatments. The increase in annual grass density and cover has largely been under cut trees and around old litter zones.

Annual Forbs - In all years annual forb density has been significantly greater in woodland versus cut plots (Table 5). Annual forb canopy cover was greater in cut plots in 1993 and 1998 indicating larger individual plant size.

Table 6. Grazed Plots - Understory biomass (lb/ac) in cut and woodland plots (1992-93 and 1996). Different letters denote significant treatment differences for each species group in each year.

Species Group	Treatment	1992	1993	1996[2]
Sandberg Bluegrass	Cut	11.5 ± 1.6 a	86.5 ± 6.2 b	N.A.[3]
	Woodland	12.7 ± 1.9 a	22.3 ± 3.6 a	
Perennial Grasses	Cut	17.2 ± 2.0 d	106.0 ± 16.0 d	N. A.
	Woodland	3.7 ± 2.0 c	5.6 ± 2.0 c	
Perennial Forbs	Cut	9.1 ± 1.2 h	25.8 ± 4.5 h	N. A.
	Woodland	4.6 ± 1.2 g	5.8 ± 0.8 g	
Annual Grasses	Cut	2.2 ± 1.3 s	9.5 ± 3.3 s	N. A.
	Woodland	0.2 ± 0.2 r	1.1 ± 0.8 r	
Annual Forbs	Cut	N. M.[4]	N. M.	N. A.
	Woodland			
Total Biomass	Cut	39.8 ± 5.1 z	228.2 ± 19.6 z	209.0 ± 37.0 z
	Woodland	20.9 ± 4.3 y	34.8 ± 5.3 y	16.0 ± 3.5 y

[2] Biomass values in 1996 are lower as plots were grazed in the spring (May 1996).
[3] N. A. - not separated by species group.
[4] N. M. - not measured.

Total ground cover and biomass - In 1998, ground cover in the cut treatment was 1.5 times greater in the cut versus woodland treatment. Ground cover totaled 44.6% (juniper debris and litter, 27%; interspace herbaceous canopy cover and litter, 16%; shrub cover, 1.6%) in cut plots compared to 33% in the uncut treatment (juniper trees and litter, 29%; interspace herbaceous canopy cover, 4.0%). It is important to note that interspace cover was over 5 times greater in cut versus woodland plots. Total understory biomass (Table 6) was greater in all years measured in cut versus woodland plots. Results show no increase in biomass in cut plots from 1993 to 1996. This is because in 1996 plots were grazed in the spring. Biomass in 1996 is only a measure of regrowth after grazing, not the actual production capability of the site.

DISCUSSION

Post-cutting Succession

Understory responses from 1991 to 1998 in grazed and ungrazed cut and woodland plots showed similar understory response trends. Unless noted, discussion will focus on cut versus woodland responses.

Understory plants in cut plots responded to juniper removal with increased productivity, plant density, and cover (basal and canopy). Removal of juniper reduces belowground competition and increases availability of soil water and nutrients to understory species, which appears to explain understory response after cutting (Bates et al. 1999, Bates 1996).

Species composition after cutting was largely composed of plants present on the site prior to juniper cutting. Thus, initial floristics of a site may be used to predict the general character of post-cutting understory response. However, making quantitative predictions of plant response are more difficult because of several unknowns, including status of seed bank reserves, time since treatment, and post-treatment climate conditions. Sandberg bluegrass and other perennial grasses dominated the understory the first three years of the study. Since 1995 succession in cut plots has continued to be dominated by the perennial bunchgrasses though we have recorded a substantial increase in density, biomass, and cover of annual grasses (cheatgrass and Japanese brome). Bluegrass cover and biomass have declined since 1993. Perennial bunchgrasses and annual grasses seem to have supplanted bluegrass in cut plots.

Until recently, succession in treated woodlands was assumed to progress through a standard pattern beginning with several years of annual dominance followed by a period of perennial grass and forb primacy. This study tends to refute that successional model. Results from our study support the multiple entrance point model (Everett 1987). In our study, pretreatment densities of perennial bunchgrass species (about 2-3 plants 10 ft^{-2}) were sufficient for bunchgrasses to recover and dominate the understory component after cutting. Whether this density value is indicative of an adequate amount of perennial grass to recover on similar sites has not been adequately tested. The results also have shown that annual grasses can become zonal dominates (under cut trees and old canopy areas) several years after removal of the overstory (Bates et al. 1998).

The increase in annual grass in 1996-1998 is attributed to two factors. First, the past several falls and winters (1995-1996, 1996-1997, and 1997-1998) have been relatively warm and wet, ideal conditions for annual grass establishment and growth. Second, the preferential establishment of annual grass under juniper debris and around the old canopy zones suggested that these areas provided good seedbeds for annual grasses and may provide other benefits such as greater soil nutrient or water availability. Annuals tend to respond well to areas of high nutrient availability. We did not measure increases in available soil N in debris or canopy zones in 1992 and 1993 (Bates 1996).

However, by 1996 juniper needles had fallen off cut trees and needle litter on the ground was being incorporated into the soil by decomposition. We suspect that needle litter decomposition resulted in greater soil nutrient release by 1996. Soil moisture levels were determined to be significantly greater under cut juniper debris than in the interspaces (Bates et al. 1998).

Though cover of perennial grasses was not high in debris and canopy zones, there has been a substantial increase in perennial grass density in these locations even under competition with cheatgrass (unpublished file data). It will be interesting to see if overtime perennial grasses will begin competing more effectively with annual grasses in these locations.

An unexpected result was the higher densities of annual forbs in woodlands compared to cut plots (Table 3 and 5). We are unsure why annual forb (primarily pale alyssum) densities were lower in the cut versus woodland plots, considering that soil moisture availability was greater in the cut treatment. Changes in micro-climate, which was not tested, could have negatively affected germination and establishment of annual forbs in cut plots.

In woodland plots, the limited understory biomass response and decrease in perennial basal cover indicated that juniper interference remained strong during wet years. Other resources, such as N, may limit understory plant growth in high moisture years. Nitrogen content was less in understory plants in woodlands compared to plants in cut plots in the wet year, 1993 (Bates et al 1999). Additional research is needed to isolate competitive interactions between the understory and overstory in juniper woodlands in wet and dry years.

MANAGEMENT IMPLICATIONS

The results indicate that the restoration of woodland sites requires patience. It may take one or more years for understory species to respond to the removal of juniper, particularly in dry years. Understory response was subtle and limited in 1992, a dry year. The main increases in plant productivity and plant densities on cut plots occurred in two stages concurrent with wet periods. The first increase occurred in 1993, and the second increase occurred between 1995 and 1998. Across much of eastern Oregon, 1993 was the wettest year on record. Growing season moisture conditions prevailing in 1995-98 were conducive to increased productivity and seedling establishment in cut plots.

However, limited understory response the first year after cutting may not be unusual even in wet years. Studies in pinyon-juniper woodlands have shown delays of 1 to several years before the understory fully responds to removal of tree interference under favorable growing conditions (Barney and Frischknecht 1974, Everett and Ward 1984). It takes time for existing plants to grow larger and new plants to become established.

The increase in herbaceous and litter ground cover in cut plot interspaces is important from a hydrologic standpoint. Research in New Mexico suggests that herbaceous ground cover is more effective at reducing erosion rates than is cover of woody vegetation (Wilcox and Breshears 1994). In the western juniper system, Buckhouse and Mattison (1980) measured greater soil erosion potential in woodland dominated sites than in sagebrush grasslands.

Grazing management on cut sites requires careful consideration. The site used in this study probably requires rest or deferment during the first few growing seasons to provide plants the opportunity to produce seed and permit seedling establishment. Grazing in late summer and fall may be permissible as plants are largely dormant during this time. Grazing (unplanned) on 4 cut plots during late summer and early fall in 1992 and 1993 did not retard understory recovery on these plots.

Comparisons between cut grazed and cut ungrazed plots were not made because sampling did not coincide in the same years since 1993. Plot differences prior to cutting for understory parameters also makes comparisons difficult between cut grazed and cut ungrazed plots. For example, grazed plots had lower perennial basal cover prior to cutting in 1991 than did ungrazed plots. Consequently, there appear to be 3 points that can be developed from grazed and ungrazed cut plot results. *First*, increases in annual grass density, cover, and biomass since 1994 have occurred under both grazed and ungrazed conditions. Removal of grazing did not prevent annual grass development. *Second*, cut ungrazed plots appeared to have had greater increases in perennial grass density. In cut ungrazed plots perennial grass density increased by 575% between 1991and 1997 while in cut grazed plots perennial grass density increased by 333% between 1991 and 1998. *Third*, in both cut ungrazed and cut grazed plots, Sandberg bluegrass cover and biomass had decreased since 1993. Cover and biomass of Sandberg is now similar to levels measured in woodlands. To develop a better understanding of grazing impacts on this site we are currently developing a more controlled grazing/no grazing study on all treatment plots. A fence has been built through the center of all cut and woodland grazed and ungrazed plots (cut and woodland). Half of each plot will be left permanently ungrazed and the other half will be grazed using various prescription grazing criteria. This study is to start in spring 1999.

On our site there was a positive correlation between juniper debris and annual grass cover, density, and biomass. Removal of cut junipers may be an option to reduce the amount of annual grass that may become established under juniper debris. Mechanical removal of debris would add an additional cost to a project and is not cost effective. If juniper becomes profitable as a wood product on a large scale then debris management would not pose a significant concern on cut sites. Burning is also effective at removing juniper debris. However burning debris the first few years after cutting has been observed to generate severe heat fluxes that may kill perennial grasses under and adjacent to debris. Therefore it has been recommend that burning of debris be deferred until fuel loads are diminished by the decomposition of juniper litter, usually a minimum of 5-10 years.

Several other reasons tend to support the argument that burning of debris should be deferred. Results from this study indicate that debris sites are preferred zones for perennial grass establishment even with fairly stiff competition from annual grasses. Burning 5, 10, or 15 years post-cutting still provides land managers plenty of time to remove seedling junipers that become established after treatment. We are currently testing to determine if debris burning can be done in the first year or two post-cutting with the twin objectives of maintaining the perennial grass component and reducing annual grass establishment sites. Burns are scheduled under very specific conditions: perennial grasses must be dormant (fall and winter) and there must be high soil and litter moisture content to reduce heat flux into the soil. Results from this study are several years away.

Cutting of trees on sites similar to the one used in this study will increase forage production and quality, improve watershed characteristics through increased ground cover, and increase plant diversity (Bates et al. 1999). However, variability in site characteristics (soils, aspect, elevation, understory composition) across the western juniper ecosystem will also influence understory response to juniper control. Additional research is required to develop models which assist land managers in predicting understory response and successional pathways after cutting trees in a variety of woodland dominated sites. Because cutting is an expensive management alternative, cutting treatments should only be applied to areas where a good understory response would be expected. Successional models would assist resource managers in targeting those areas where desirable understory vegetation may be successfully restored by juniper cutting.

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UNDERSTORY VEGETATION CHANGE FOLLOWING CUTTING OF WESTERN JUNIPER IN GRANT COUNTY, OREGON

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SUMMARY

Plant response to juniper tree cutting was compared with adjacent uncut plots in Grant County, Oregon. The study area was a closed juniper woodland with overstory trees ranging in age between 70 and 90 years. Measurements were recorded before treatment in 1992 and after treatment in 1993 and 1995. Response of Idaho fescue and bluebunch wheatgrass were minimal and were probably due to initial densities being less than one plant per 10 ft^2. However, squirreltail and perennial forbs, which had initial densities of near or greater than one plant per 10 ft^2, increased following cutting. Densities of all shrub species before treatment were too sparse to show a measurable response in the third post-treatment year.

INTRODUCTION

The decision to remove an overstory of western juniper (*Juniperus occidentalis*) on any site or landscape should be based on a number of site factors, including quantitative data on age/size classes of trees, degree of stand closure, soil characteristics, and available understory plant resources. Understory species composition and density are dominant factors determining vegetation response following cutting. In addition, the rate of response will also be controlled by a multitude of abiotic and biotic factors. However, there are few studies to serve as guides for prediction of understory vegetation response. Of particular concern are thresholds of desirable plant species abundance, below which there is little response after treatment. When desired species are incapable of responding, a vegetation state dominated by undesirable annuals or alien species may develop.

This study provides useful guidelines for expected understory response after cutting of western juniper. Although responses reported only extend across three growing seasons immediately following treatment, the rate of change typically slows after three years as the sites resources are largely reallocated in this time frame.

STUDY AREA

The study area was located in Grant County near Mount Vernon. The site is dissected by a second-order drainage generally flowing south-westerly. The upper elevation of the study was approximately 4300 ft elevation, against the ponderosa pine zone, and extended downward to approximately 3200 ft elevation. The two opposing sides of the drainage have general aspects of NW and SE respectively. Occasionally, juvenile ponderosa pines were found on those plots with a NW aspect but not on plots of the opposing SE aspect. Regional precipitation during the study was below average in 1992, 1993 was wet, while 1994 and 1995 were near average years.

The NW aspect has been mapped (USDA, Soil Con. Ser. 1981) as a Snell-Anatone complex. These are both stony loams ranging from 11 in. deep (Anatone) to 23 in. deep (Snell) overlying basalt. The soils on the SE aspect were mapped as a Lickskillet-Rock outcrop complex. The Lickskillet series, which strongly dominates this particular area, is an extremely stony loam about 15 in. deep to basalt.

The plant community before treatment was a western juniper overstory and a Sandberg bluegrass-annual understory. Given the elevation, amount of precipitation (estimated to be roughly 16 in.), soils, remnant vegetation, and vegetation of similar sites, the historic plant community on the NW aspect was most likely dominated by Idaho fescue. Estimated composition based on production for the historic community is Idaho fescue 60%, bluebunch wheatgrass 10%, mountain big sagebrush 10%, perennial forbs 10%, and Sandberg bluegrass, other bluegrasses, squirreltail, junegrass, and antelope bitterbrush all at about 2% each. The SW aspect was likely a bluebunch wheatgrass community with the following expected historic composition of bluebunch wheatgrass 50%, Idaho fescue 10%, Thurber needlegrass 5%, mountain big sagebrush 5%, Sandberg bluegrass 3%, and antelope bitterbrush 2%.

METHODS

Vegetation Measurement

Three paired plots were located on each side of the main drainage for a total of 12 plots (6 plots on each aspect). One of the plots within each of the six pairs was randomly selected for cutting. Western juniper trees on the selected plots were cut by chainsaw during the fall of 1992 and winter of 1993 and left whole on the ground. Plots varied in size, ranging from 12 to 25 acres. Vegetation measurements included understory cover (percent of canopy estimate) and density (number of individuals counted in meter square sample plots) in the summer of 1992 before treatment and again in the summers of 1993 and 1995 after treatment. Sample plots were located along ten random lines in each of the 12 plots. Western juniper cover was measured in the summer of 1992 with the line intercept method along three 60-m transects in each of the 12 plots. Tree density was measured in three 6x60-m belt transects in each plot. Vegetation data reported here includes data from the 1992 pre-treatment and the third year post-treatment, 1995.

Wildlife

Composition and abundance of small mammals and birds were measured in the three plot pairs located on the NW aspect during three summers following treatment. These results are reported in two papers presented in this publication.

RESULTS

Pretreatment Plant Measurements

In 1992 we estimated western juniper cover at 42% across all plots with no differences between plots. This was a closed, even-aged woodland with dominant overstory trees varying between 70 and 95 years in age. Suppressed sapling trees (< 12 ft tall) in the understory ranged in age from 45 to 60 years. Recruitment of young trees was rare, with juvenile tree density < 15/

acre and displaying suppressed leader growth. Fruit production across the stand was scarce. Tree density was 122/acre across the NW aspect plot and 102/acre across the SE aspect plots.

Bare ground and western juniper accounted for the majority of cover across all plots (Fig. 1). Litter cover was nearly 30%. However, over 80% of the litter was juniper needle mat located beneath the tree canopies. Total herbaceous plant cover was about 10% and not different between the two aspects. However, components of the herbaceous vegetation shifted in importance with aspect. Sandberg bluegrass, other perennial grasses, and perennial forbs were important components on the NW aspect while Sandberg bluegrass and annual forbs were most important on the SE aspect (Fig. 2). Sandberg bluegrass consistently made up over 2/3s of the perennial grass component on both aspects. Shrubs were noticeably scarce and seldom encountered in sampling.

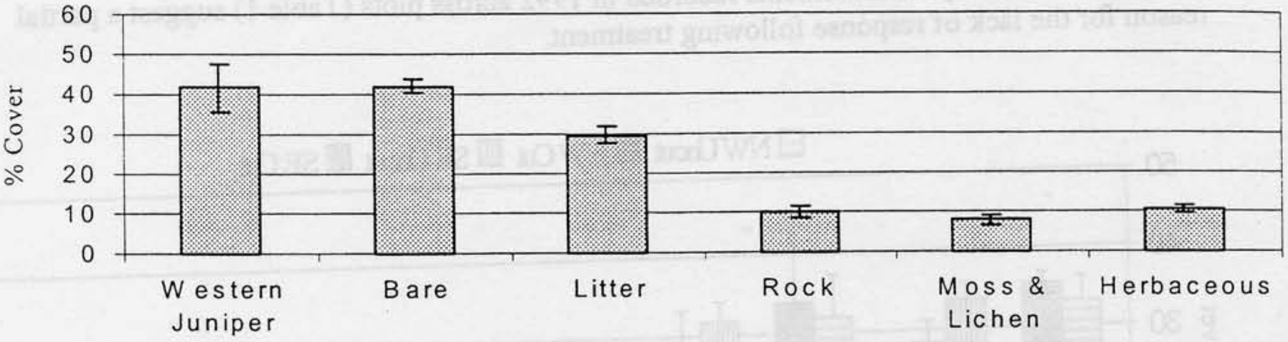

Figure 1. Pretreatment surface cover (%) in major catagories.

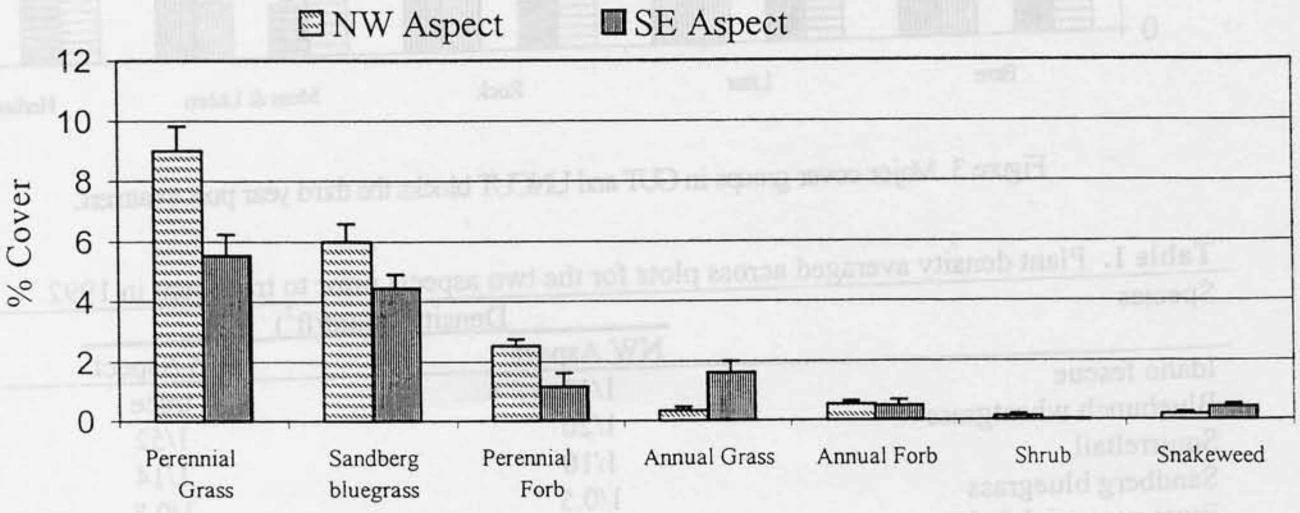

Figure 2. Pretreatment ground cover (%) by major plant groups and species.

Post-treatment Plant Measurements

Total perennial grass cover on the NW aspect and Sandberg bluegrass cover on both aspects declined on the uncut plots between 1992 and 1995. Other components were not different. Cut plots showed less moss and lichen cover and greater herbaceous cover than uncut plots for the NW aspect, however, other components were not different (Fig 3). As to the herbaceous components, cut plots on the NW aspect resulted in a higher cover of perennial grass, perennial forb, annual grass, annual forb, and snakeweed than did the uncut plots (Fig. 4). Cut plots on the SE aspect resulted in less Sandberg bluegrass but greater annual grass and snakeweed cover compared to uncut plots. Sandberg bluegrass contribution to perennial grass cover had dropped to approximately 30% in 1995 on cut plots.

Cover response by herbaceous perennial components to cutting of western juniper was not spectacular. Density measurements recorded in 1992 across plots (Table 1) suggest a partial reason for the lack of response following treatment.

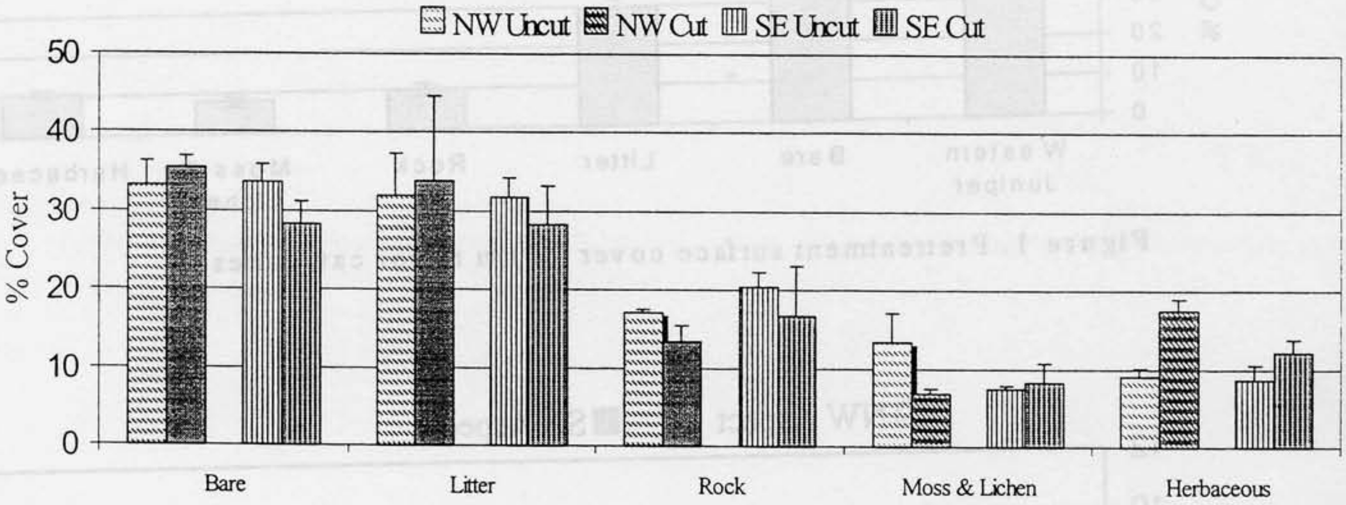

Figure 3. Major cover groups in CUT and UNCUT blocks the third year post treatment.

Table 1. Plant density averaged across plots for the two aspects prior to treatment in 1992.

Species	Density (plant/ft^2) NW Aspect	Density (plant/ft^2) SE Aspect
Idaho fescue	1/13	trace
Bluebunch wheatgrass	1/20	1/52
Squirreltail	1/10	1/14
Sandberg bluegrass	1/0.5	1/0.8
Erect perennial forbs	1/3	1/8

In uncut plots in 1995, density of Sandberg bluegrass plants had declined 40% on the NW aspect while erect perennial forbs had nearly doubled on both aspects. Annual grass density had also increased. Cut plots showed a 2 to 3 fold increase in squirreltail and a 3 to 4 fold increase in erect perennial forb densities on the two aspects. Mat forming perennial forbs also increased 2 to 3 fold. Although individuals of mat formers are difficult to separate, new juveniles were particularly abundant for western yarrow. Sandberg bluegrass showed a 50% decline in density on both aspects. Annual grass density tended to increase 1 to 2 orders of magnitude.

CONCLUSIONS

Juniper establishment peaked between 1900 and 1920 with little tree recruitment occurring after 1945 across the study site. Suppressed leader growth on understory trees suggested the woodland was closed (trees fully occupied the site, see Table 2 in "Understory Dynamics…", in this publication). The relatively low response by Idaho fescue and bluebunch wheatgrass to the removal of western juniper is likely due to a combination of insufficient numbers of individuals as well as inadequate time to respond. Measurable responses were not evident for either numbers of individuals or for cover. Numbers less than one per 10 ft^2 may represent a threshold in measurable plant response and additionally may be too few to produce sufficient seed to repopulate the area. Squirreltail, however, can be a prolific seed producer compared to Idaho fescue and bluebunch wheatgrass, particularly following disturbance. Post-treatment squirreltail numbers increased following juniper treatment. This species had an initial density near 1 per 10 ft^2, which had increased on cut plots in 1995 to 1 per 4 ft^2 on the NW aspect and to 1 per 5 ft^2 on the SW aspect. Perennial forbs as a group had densities greater than 1 per 10 ft^2 before treatment and these also increased significantly. Shrubs of all species were too sparse to show measurable increases in cover, and insufficient time had probably elapsed for new individuals to appear. However, snakeweed, which is a half/shrub increased in numbers and cover after cutting.

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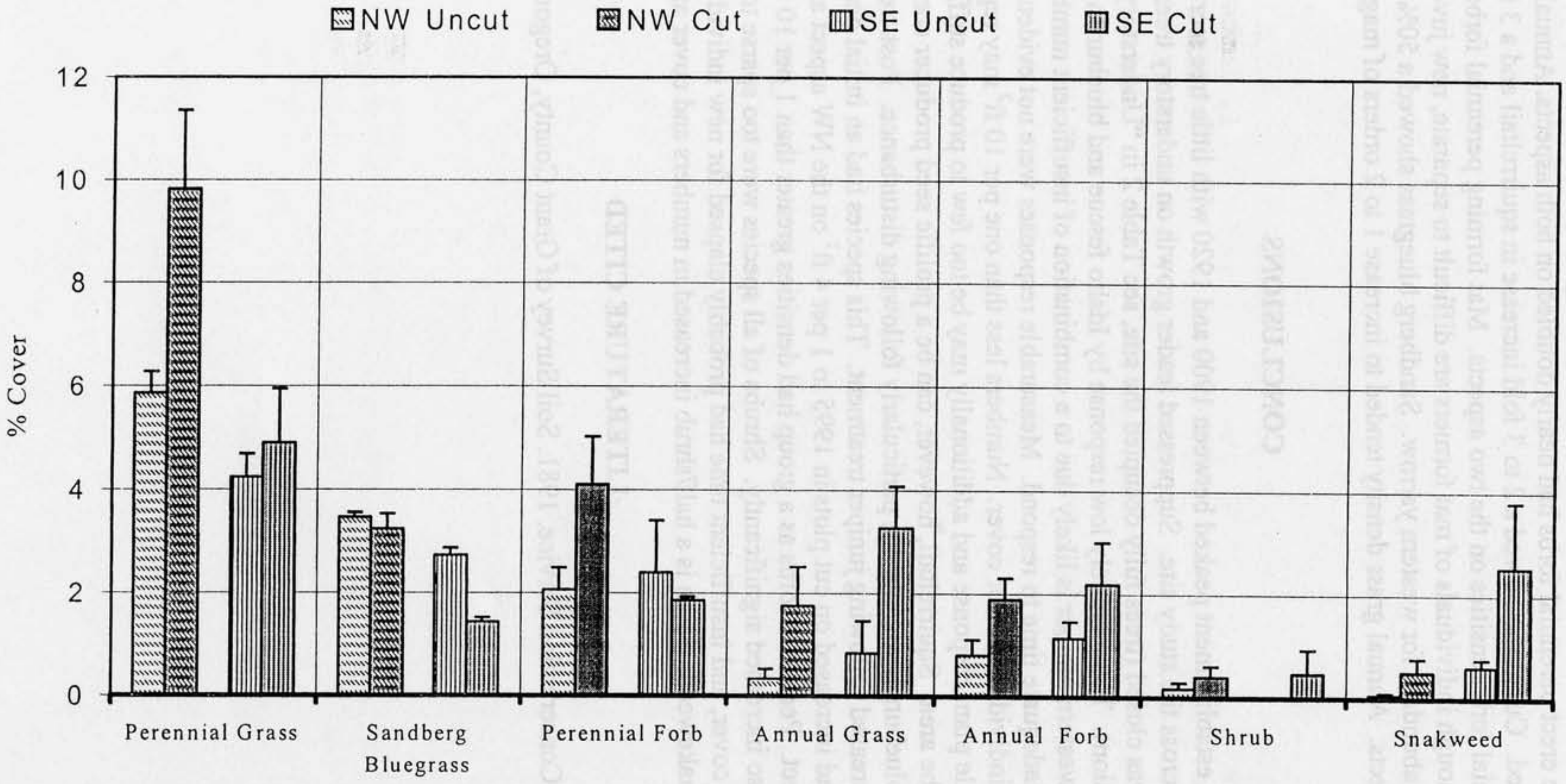

Figure 4. Plant cover by functional groups and species on UNCUT and CUT blocks the third year post treatment.

PERSISTENCE OF IDAHO FESCUE ON DEGRADED JUNIPER/SAGEBRUSH/STEPPE IN CENTRAL OREGON

Paul Doescher, Jay Goodwin, Lee Eddleman, Richard Miller, Ray Jaindl and Mohamed Nasri

INTRODUCTION

Over the past 150 years, abusive land management practices coupled with introductions of exotic weeds, as well as increases in shrubby plant species, have led to a dramatic decline in abundance of native perennial grasses. Efforts to reestablish native grasses on degraded lands has met with limited success, often due to lack of adapted, available, and competitive cultivars. There is a growing need for adapted cultivars of native grasses; however, their usefulness in restoration efforts will be determined by their ability to tolerate such factors as moderate grazing intensities, competitive pressure from both annual weeds and shrubby species, and their ability to exist under the current and future climatic regimes of central Oregon.

The purpose of this report is to summarize the findings of a long-term research project which sought to determine reasons why remnant populations of Idaho fescue (*Festuca idahoensis* Elmer.) persisted on disturbed central Oregon rangelands. We had observed that individual plants of Idaho fescue were present on certain degraded, central Oregon, sagebrush steppe sites despite these areas being accessible to heavy grazing pressures, invasion by alien weeds such as cheatgrass (*Bromus tectorum* L.), and increased densities of western juniper (*Juniperus occidentalis*) and sagebrush (*Artemisia tridentata*) (Mack 1981, Miller and Wigand 1994, Burkhardt and Tisdale 1976, Galbraith and Anderson 1971). Hypotheses designed to test the observed persistence of remnant Idaho fescue plants on degraded sites:

1) These populations represented grazing-tolerant ecotypes of Idaho fescue.
2) These populations were composed of the most competitive ecotypes of Idaho fescue.
3) These populations represented the most stress-tolerant ecotypes of the original populations.

A series of research trials were performed to test these hypotheses.

Research Sites

Plant materials of Idaho fescue, including seeds and transplants, were collected from 5 sites in central Oregon (Fig. 1). Four sites were classified as disturbed, and one site was considered undisturbed. The undisturbed site had been previously described by Driscoll (1964) as "The Island", and is considered a pristine location. Sites were selected for their similarity: 3 of the disturbed sites had a high canopy coverage of juniper, and the undisturbed site had only a trace amount not measured in our field sampling (Table 1). Given the history of livestock production in central Oregon (Galbraith and Anderson 1971), and the palatability of Idaho fescue (Vavra and Sneva 1978, Drast and Havstad 1987), the disturbed sites had likely experienced a long-term history of grazing by livestock. Two research directions were emphasized: 1) assessment of plant response to simulated grazing trials and 2) examination of seedling growth in competitive environments with cheatgrass.

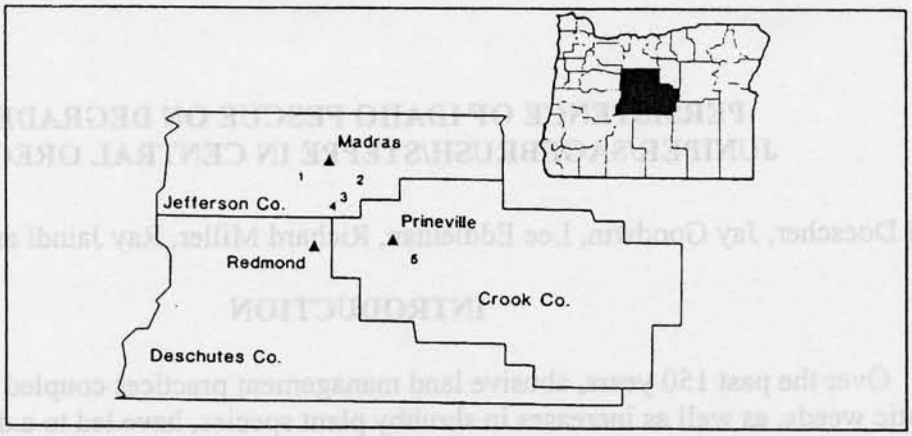

Figure 1. Location of plant material collection sites in central Oregon: 1 is the undisturbed site, 2-5 are the disturbed sites.

Table 1. Topographic edaphic and vegetation characteristics of research sites in central Oregon.

Site Location	Elev. (m)	Slope of Aspect	Soil Series	Idaho fescue cover %	Cheatgrass cover %	Sagebrush cover %	Western juniper cover %
Island (undisturbed)	740	plateau	Agency-Meadow Complex	6	---	2	---
McCoin	1015	35% N	Prag	8	<1	---	41
Blanchard	1185	5 % N	Gribble	7	11	<1	7
Lone Pine	970	40 % NW	Curant-Tub Complex	2	<1	---	28
Combs Flats	1050	20 % N	Lookout	1	9	5	19

[1] adapted from Goodwin et al. (1999).

SIMULATED GRAZING RESEARCH[1]

Experimental Approach

This experiment was designed to test whether Idaho fescue from the disturbed sites exhibited greater tolerance to grazing than Idaho fescue from the undisturbed site. Experiments were conducted in 1989 and 1990, using plants transplanted to a common garden established at the Powell Butte location of the Central Oregon Experiment Station.

Defoliation treatments were based upon plant phenology and included: 1) no clipping (control), 2) clipping when plants were in the 3-4 leaf stage (vegetative stage), 3) clipping when plants were beginning to exert inflorescences (boot stage), and 4) clipping when plants were flowering (complete inflorescence emergence). Plants were clipped to a height of 5 cm (2 in.) above ground level. Material clipped during each treatment period was dried for 48 hours at 50 °C and weighed to estimate standing crop removed at time of clippings. In mid-August, all plants were clipped to a 5 cm height, dried and weighed to assess end-of-year biomass. Basal area of plants was also measured at this time; biomass was standardized on a basal area basis. Plant height was also recorded during the second growing season.

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[1] See Jaindl et al. 1994 for more details

Results and Discussion

Significant differences in plant height and biomass were found for unclipped plants between the disturbed-site populations and the undisturbed population. Plants from the disturbed-site populations were shorter and produced less biomass per unit of basal area than plants from the undisturbed site. These results are consistent with findings from other research which have shown that long-term exposure to heavy grazing results in the genetic shift of plants to lower statured plants (Detling and Painter 1987). However, when comparing biomass of plants from all collection areas averaged across defoliation treatments, no significant differences in response were noted among the collections. Defoliation did not result in reduction in yearly aboveground biomass, regardless of time during which defoliation was applied (Fig. 2). In 1991, defoliation at the boot and anthesis stages resulted in an increased yearly biomass accumulation.

These results suggest that grazing history had no consistent influence on the response of Idaho fescue to defoliation. Idaho fescue appears to be a species with relatively high tolerance to grazing. Ability to regrow rapidly and reestablish above ground biomass is a mechanism often cited as a characteristic of defoliation tolerant bunchgrass (Caldwell et al. 1981). Unlike bluebunch wheatgrass (*Agropyron spicatum* (Pursch.) Scribn. & Smith), which has poor regrowth characteristics following heavy defoliation (Caldwell et al. 1981), Idaho fescue recovered well following defoliation.

Results of this study suggest that survival of Idaho fescue in grazed areas may not be a result of a response to grazing in terms of regrowth after defoliation or time of phenologic development, but a result of changes in plant stature. These results are consistent with findings of McNaughton (1984). Grazing may have resulted in selective survival of shorter statured plants, which would be less exposed to the grazing animal.

An offshoot of the results of the defoliation component of this study is the speculation that Idaho fescue may exhibit grazing tolerant characteristics even though it evolved in a biome in which large herbivore numbers did not exist (Mack and Thompson 1982). Its weakness may be its high palatability, which makes it prone to repeated defoliation during its active growing periods.

GERMINATION AND COMPETITION RESEARCH[2]

Our initial predictions were that Idaho fescue germination characteristics and seedling competitive abilities would be better for disturbed-site populations than the undisturbed populations. Specifically, we felt seeds from the undisturbed site populations would germinate faster and be less sensitive to moisture stress than seeds collected from the disturbed site. In addition, we also speculated that seedlings from the disturbed sites would exhibit growth characteristics that showed them to be better competitors with cheatgrass than seedlings from the undisturbed site.

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[2] See Goodwin et al. 1999 and Nasri and Doescher 1995 for further details

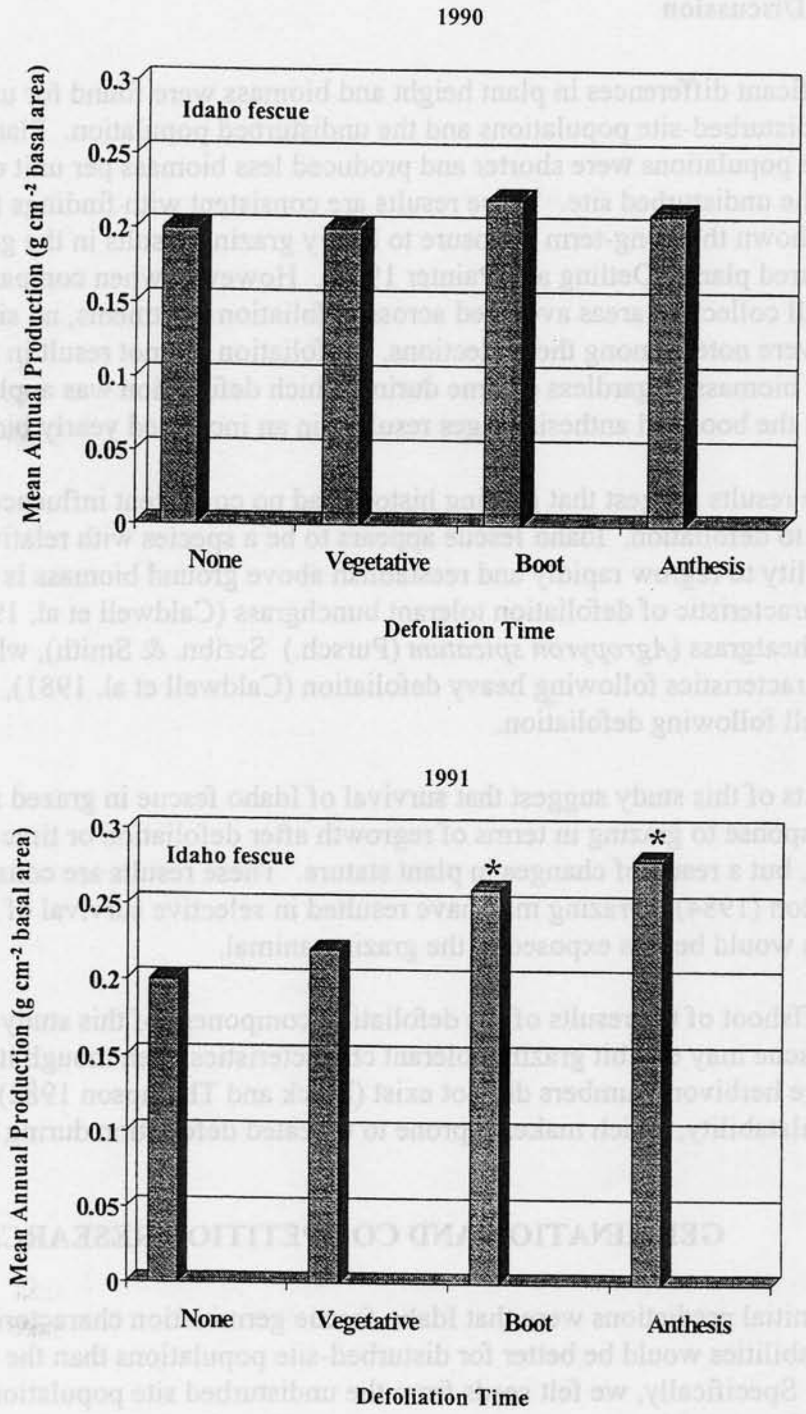

Figure 2. Plant species response to defoliation. Mean relative annual production of Idaho fescue collections from central Oregon grown in a common garden. Values represent the average of all collections: defoliation occurred at vegetative, boot, or anthesis stages of growth. Significant difference in values noted with an *. From Jaindl et al. (1994).

Experimental Approach

Seeds of Idaho fescue and cheatgrass were brought back to Corvallis, Oregon, for evaluation under greenhouse and environmental chamber evaluations. Specific research trials performed included:

1) Evaluation of germination characteristics to various temperature and moisture stress environments.
2) Evaluation of seedling root growth characteristics.
3) Evaluation of Idaho fescue seedlings in competition with varying densities of cheatgrass seedlings.

Results and Discussion

Results from the germination trials indicated that Idaho fescue from the disturbed sites did not represent superior ecotypes in comparison to the undisturbed site. A greater proportion of seeds from the undisturbed site germinated; they germinated faster and were no more sensitive to simulated water stress than were seeds from the disturbed sites (Fig. 3).

In competition experiments with cheatgrass, results discounted the hypothesis of greater competitiveness in disturbed site Idaho fescue seedlings. In one experiment (Goodwin et al. 1999) where seedlings of Idaho fescue were grown with varying proportions of cheatgrass seedlings, Idaho fescue from the undisturbed site exhibited a higher above ground growth and greater below ground growth than seedlings from the disturbed sites (Table 2). For instance, seedlings from the undisturbed site produced approximately 3.5 times the root length of disturbed site seedlings. However, in a related experiment (Nasri and Doescher 1995) growth of Idaho fescue from the undisturbed site without competition from cheatgrass produced 1.7 times as much shoot biomass as disturbed site populations. But, addition of 5 cheatgrass seedlings around the Idaho fescue seedlings reduced shoot biomass of the undisturbed site seedlings by 35%. No decrease in biomass was found for disturbed site seedlings in comparison to their response in the absence of cheatgrass competition.

Root growth response was also different among the Idaho fescue collections. In a comparison of root growth characteristics of Idaho fescue and cheatgrass grown in root tubes, it was found that Idaho fescue from the undisturbed site had roughly 3 times greater root length than undisturbed sites seedlings (Goodwin et al. 1999). However, one major difference was found among the Idaho fescue collections. Seedlings from the undisturbed sites concentrated root growth in the upper soil profile, while disturbed site seedlings had a greater concentration of root biomass in the lower soil depth. Cheatgrass had both greater root biomass, root length, and root mass at all depths compared to the Idaho fescue seedlings.

Table 2. Size and allocation patterns in Idaho fescue seedlings from disturbed and undisturbed sites, and in cheatgrass seedlings.[1]

	Idaho fescue				Cheatgrass	
	Undisturbed[1]		Disturbed			
	Mean	SE	Mean	SE	Mean	Se
Emergence (days)	7.00a	1.31	11.00b	0.66	4.80	1.31
Total Root Length (cm)	963.32a	158.10	273.09b	72.54	16348.88	769.43
Total Leaf Area (cm^2)	11.66a	1.47	3.44b	0.67	125.42	6.27
Root Biomass (mg)	27.85a	3.44	8.57b	1.58	890.13	72.57
Shoot Biomass (mg)	71.15a	6.82	17.76b	3.07	707.10	43.51
Root Weight Ratio[4]	0.29a	0.05	0.33a	0.02	0.55	0.03

Within a row, different letters indicate significant differences (α=0.05)
[1] n=4
[2] n=19
[3] n=5
[4] Root weight ratio = g root/g root+shoot
[1] from Goodwin et al. 1999

INTERPRETATION AND CONCLUSION

Given that greater competitive ability does not explain the persistence of Idaho fescue on degraded sites, we reject Hypothesis 2. Instead, Hypothesis 3 and in part Hypothesis 1 offer the most likely explanations for the persistence of remnant plants. Our results suggest the following:

1) That although all Idaho fescue populations exhibited a high degree of defoliation tolerance (as shown by high regrowth response), there has been a genetic shift to shorter statured plants. Shorter statured plants would be less exposed to grazing animals, which would allow them to maintain a greater proportion of their leaf area available for photosynthesis.

2) That disturbed site Idaho fescue have persisted on grazed and invaded sagebrush-steppe because of greater tolerance to moisture stress. A slow, largely invariable rate of growth typically accompanies stress-tolerance, whereas the capacity for rapid production of roots and leaves in response to transient resource enrichment (such as high soil moisture) confers competitive ability (Grime 1979).

We believe Idaho fescue has persisted on grazed and invaded sagebrush-steppe because of an ability to withstand relatively high levels of grazing, drought tolerance to moisture stress such as that imposed by drought or competition with other species, but not greater competitive ability. Disturbed-site Idaho fescue populations likely represent stress tolerant ecotypes resulting from intensified selection pressures, from shrub increases and introductions of alien species. Our germination results suggest that recruitment of Idaho fescue seedlings is infrequent on invaded sites, and persistence of these populations requires longevity of established genotypes or recruitment of new individuals or both. Seeds remain important to long-term survival and evolution of the populations because of new genotypes generated through sexual reproduction. However, between episodes of seedling recruitment, clone formation by adapted genotypes is, we believe, key to the persistence of Idaho fescue on invaded sagebrush-steppe. Proper grazing management is also essential to long-term survival of Idaho fescue. Understanding the biology of Idaho fescue and other native perennial grasses provides a foundation for proper rangeland management, having implications that extend from sustainable grazing management to conservation and restoration.

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OLD GROWTH WESTERN JUNIPER WOODLANDS[1]

Rick Miller, Wendy Waichler, and Paul Doescher

SUMMARY

Researchers and resource managers in forestry, rangeland, and ecology have generally overlooked semiarid old-growth woodlands. These ancient woodlands have some of the oldest trees in the Intermountain region, exceeding ages of 1,000 years. Typically, old-growth are structurally more complex than postsettlement woodlands, adding biological diversity to the landscape and providing an important source of habitat for many organisms. Mapping and inventorying old-growth woodlands are extremely important in developing management and land-use plans.

INTRODUCTION

Old-growth juniper and pinyon woodlands in the West generally do not fit the typical image most people have of old-growth coniferous forests. In a recent symposium in the southwest, Swetnam and Brown (1992) stated, "Many peoples' image of old-growth are the stately monarch trees with shafts of sunlight streaming down through tall, dense canopies. However, in the southwest, many of the old-growth stands do not fit this stereotype." Some of the oldest stands throughout the Intermountain West are low statured, open, semiarid woodlands composed of such species as bristlecone, limber pine, juniper, and pinyon. Old woodlands usually differ in structure and function from postsettlement woodlands thus adding diversity at the community and landscape levels. Although considerable research has been conducted in old-growth for other conifer species, work addressing old-growth in juniper and pinyon woodlands is very limited. In addition, the concern over the rapid expansion of juniper and pinyon woodlands during this century has overshadowed the presence and value of these presettlement woodlands. Ancient woodlands are frequently overlooked in management plans and inventories where they are often lumped with postsettlement stands. Wildlife studies conducted in juniper or pinyon-juniper woodlands have also not separated post- from presettlement stands. In addition, there have been occasions when mature western juniper woodlands have been misidentified as old-growth.

The intent of this paper is to describe old-growth as it relates to western juniper woodlands, summarizing their characteristics and variability, then contrasting these ancient woodlands with younger postsettlement stands.

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[1] This paper is an abridged version of a paper published In: Monsen, Stephen B.; Stevens, Richard; Tausch, Robin J.; Miller, Rick; Goodrich, Sherel. 1998. Proceedings: Ecology and management of pinyon-juniper communities within the interior west. 1997 Sept. 15-18 Provo, UT. Gen. Tech. Rep. INT-GTR-OOO. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station, with updates from ongoing EOARC research.

OLD-GROWTH: A GENERIC DESCRIPTION

Old-growth work in the northwest United States has been focused on the more mesic heavily forested areas. In the Great Basin, documentation of old-growth woodlands is almost non-existent. What is known about old-growth juniper and pinyon comes largely from anecdotal mentions in the literature, noting the occurrence of presettlement trees on rimrock, low sagebrush tablelands, and other fire resistant areas. Several attempts have been made to describe pinyon-juniper old-growth (Popp et al. 1992, Mehl 1992) but little quantitative work has been conducted in these stands.

The US Forest Service defines old-growth forests generically as ecosystems distinguished by old trees and related structural attributes. Their definition states that old-growth encompasses the later stages of stand development that typically differ from earlier stages in a variety of characteristics which may include tree size, accumulations of large dead woody material, number of canopy layers, species composition, and ecosystem function (USDA Forest Service 1993). Structural features important in characterizing old-growth in the Intermountain West vary widely across forest type, climate, site conditions, and disturbance regimes (Kaufmann et al. 1992).

OLD-GROWTH WESTERN JUNIPER WOODLANDS

Although structure, composition, and ecology of semiarid woodlands is considerably different than the wetter heavily forested areas, the principles for defining these stands, which are based on tree age, stand structure, and function, should remain the same. Nonetheless little to no information is available on stand structure, rates of mortality and the general ecology of these semiarid woodlands. Information relating old-growth woodlands to wildlife values is also limited since very few wildlife studies have described stand structure or separated old-growth from postsettlement woodlands.

Old-Growth Juniper: A Single Tree Perspective

A frequently asked question is; "What is an old-growth juniper?" One age separation frequently made is on the basis of tree establishment occurring prior to or following Eurasian settlement. In the Intermountain Northwest the rapid expansion of western juniper coincided with Eurasian settlement in the late 1860s and 1870s (Burkhardt and Tisdale 1976, Miller and Rose 1995, 1999). Woodland expansion for much of this region began in the 1870s. Based on the chronology of past events throughout the northern Great Basin, we define postsettlement trees as having established sometime after 1870, and presettlement trees establishing before 1870. However, old-growth can also be based on structural characteristics of the tree, which develop slowly over time. As juniper ages, canopy morphology shifts from cone shaped to a rounded top. As age advances the tree may also develop a combination of the following characteristics: broad nonsymmetric tops, deeply furrowed bark, twisted trunks or branches, dead branches and spike tops, large lower limbs, trunks containing narrow strips of cambium (strip-bark), hollow trunks, large trunk diameters relative to tree height, and branches covered with a bright yellow green lichen (*Letharia* sp.) in both juniper and pinyon. Tree size, particularly height, which correlates poorly with age, is dependent upon site characteristics.

Tree age within an old-growth stand is an important index in assessing the stage of old-growth woodland development. Assessing stand age also determines the rarity or uniqueness of the woodland. Individual western juniper trees can easily attain ages exceeding 1,000 years (Miller unpublished data). The oldest living western juniper tree (*Juniperus occidentalis* ssp. *occidentalis*) currently reported is just over 1,600 years old. However, many old trees cannot be aged because trunk centers are rotten.

Old-Growth Juniper: A Woodland Perspective

At the community level, old-growth juniper woodlands are best described on the basis of the presence of old trees and structural characteristics such as standing and down dead, decadent living trees, cavities, and branches covered with lichens. A single set of attributes and quantities cannot classify all stands as either old or young. Waichler (1998) recently developed a quantitative old-growth western juniper description for the aeolian sand region in Central Oregon. She stated structural characteristics that distinguished these old growth western juniper woodlands from postsettlement stands were tree growth form, standing dead, down dead, lichen growing on the dead branches, and cavities. These stands were also characterized by a relatively open overstory canopy. Old-growth woodlands in central Oregon contained a minimum density of 80 trees ha^{-1} at least 200 years old. These trees were typically 16 cm in diameter, contain less than 90% of a full canopy, and exhibited morphological characteristics of old-growth trees. However, some trees retained the postsettlement growth form with symmetrical conical canopy shapes up to 250 years. Tree canopy cover ranged between 10 and 35% with tree basal areas varying from 18 to $39 m^2 ha^{-1}$. Dead wood within the stand was primarily retained as detritus in live trees and as standing dead trees. A minimum of 10 standing dead trees/ha (25 cm diameter) and 1 large (3.3 m length and 25 cm diameter) downed piece/ha occurs.

The prehistoric record indicates the distribution and dominance of presettlement western juniper woodlands have greatly fluctuated during the past 5,000 years (Miller and Wigand 1994). Western juniper generally increased during periods of mild wet climate, declining with an increase in fire at the end of these wet periods. The pollen record indicates western juniper declined during the past 500 years before settlement. In contrast, stands that have established after the 1870s appear to be considerably more dense, have developed under different environmental conditions, and occupy more productive and deeper soil sites than these presettlement woodlands.

In Oregon, it is estimated that less than 3% of the current 5 million acres of western juniper woodlands are characterized by trees > 100 years old (USDI-BLM 1990). Although not well documented, similar proportions of western juniper old-growth are probably found in northeastern California, northwestern Nevada, and southwestern Idaho. Acreage of old-growth is not known since mapping and inventory of old-growth western juniper woodlands is limited throughout its range. In addition, the proportion of pre- and post settlement trees varies across ecological provinces.

OLD-GROWTH WESTERN JUNIPER WOODLANDS
ECOLOGICAL PROVINCES

Ecological provinces[2] (Fig. 1) provide a useful first cut to separate or classify old-growth juniper woodlands. Not only does the abundance of old-growth woodlands vary among provinces but it is likely that stand structure, composition, and probably the ecology also differ. Soils in the High Desert and Klamath ecological provinces, and the Owyhee Plateau in the Humboldt province are primarily derived from igneous rock (basalt, andesite, and ryolite). Igneous rock is also the most abundant parent material in the southwestern portion of the Snake River province, where western juniper occurs. In these provinces old-growth juniper typically grows widely spaced on shallow, rocky, heavy clay soils, or rimrock supporting limited fine fuels to carry a fire. Juniper Mountain, in Harney and Lake Counties, is a unique example of dense, old-growth woodlands growing on deep well-drained soils, which typically supports mountain big sagebrush steppe community types. Sedimentary soils, which occupy a large portion of the John Day province, support widely spaced old trees with little understory to carry fire. The aeolian sands in the Mazama and eastern edge of the High Desert provinces support the most extensive stands of old-growth western juniper woodlands.

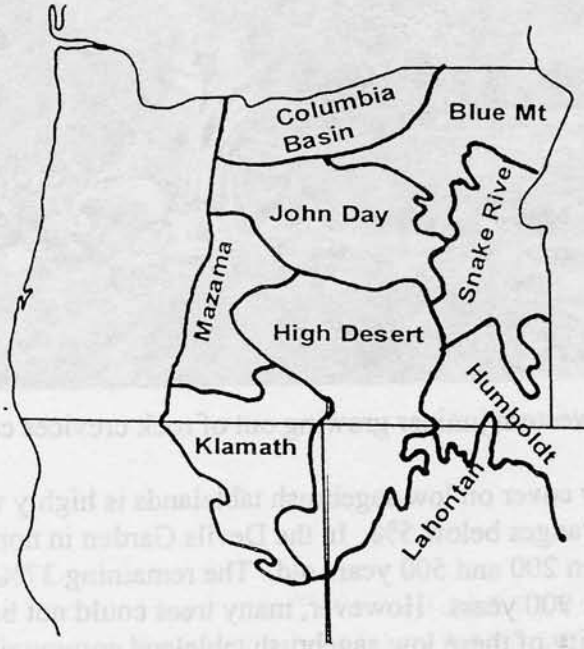

Figure 1. Ecological Provinces in eastern Oregon (derived from Anderson 1956, Cronquist et al. 1972, and Bailey 1980).

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[2] Ecological provinces are defined by floristics, plant-soil relationships, climate, topography, geology, and soils. However similarities and dissimilarities of vegetation between provinces are not always clear. Derived from Anderson 1956, Cronquist et al. 1972, and Bailey et al. 1994.

Igneous zone: High Desert, Klamath, Humboldt, and Snake River Ecological Provinces

Presettlement juniper trees are typically found on rocky surfaces or ridges (Fig.2), and low sagebrush tablelands in the High Desert, Klamath, Humboldt, and Snake River Ecological Provinces. The low sagebrush tablelands occupied by presettlement juniper trees (Fig. 3) most likely account for the greatest proportion of old-growth juniper across these provinces. Old-growth juniper probably accounts for less than 3% of the woodlands across these provinces. The dominant grass is typically Sandberg bluegrass, with Idaho fescue growing beneath the juniper tree canopies. These juniper/low sagebrush tablelands often occupy extensive flats typically with less than 5% slope, although slopes can approach 30%. The rocky, shallow heavy clay soils are primarily of igneous origin. Although soils are shallow juniper roots often penetrate the fractured bedrock.

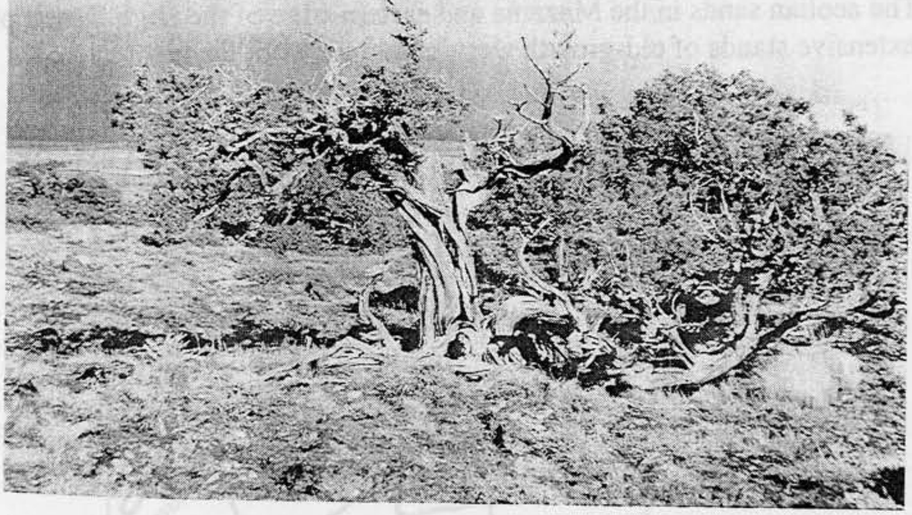

Figure 2. Ancient western juniper growing out of rock crevices covered by a few inches of soil.

Tree canopy cover on low sagebrush tablelands is highly variable and may approach 20%, but typically ranges below 5%. In the Devils Garden in northern California, 63% of trees aged ranged between 200 and 500 years old. The remaining 37% were older than 500 years with the oldest aged near 900 years. However, many trees could not be aged because of rot. Tree densities in a majority of these low sagebrush tableland communities have increased during the past 100 years (Young and Evans 1981, Miller and Rose 1995, 1999). Low presettlement tree densities in these communities can probably be attributed to limited tree establishment due to heavy clay soils, slow growth rates, and occasional fires. Occasional fires did burn across these low sagebrush Sandberg bluegrass community types with estimated return intervals of 100 to 200 years (Young and Evans 1981, Miller and Rose 1999). However, single tree lightening fires were probably more common occurrences across the juniper low sagebrush tablelands.

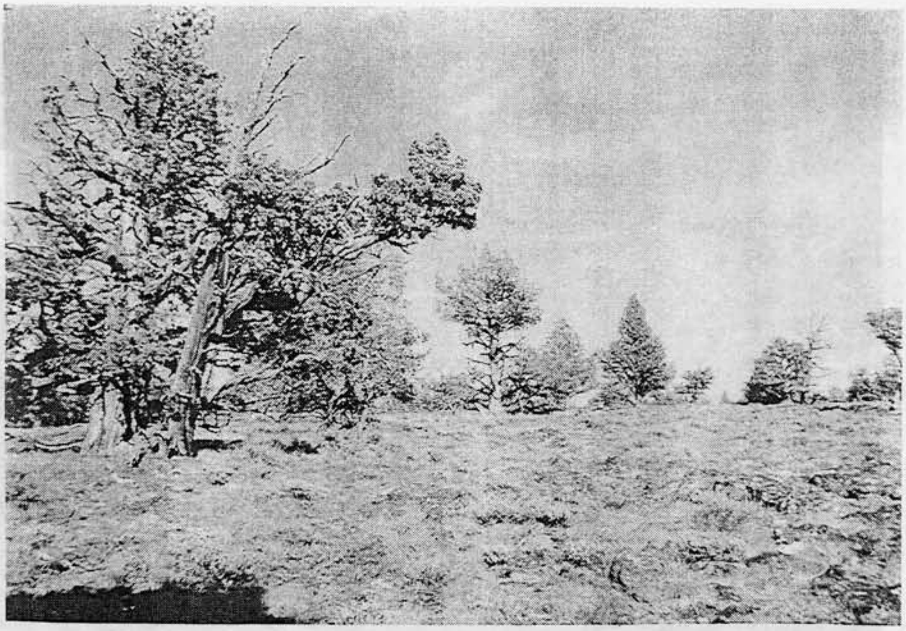

Figure 3. Juniper low sagebrush Sandberg bluegrass tablelands. Tree canopy is usually sparse and trees are widely scattered.

On the deeper igneous soils, fire limited the development of old-growth western juniper woodlands (Miller and Wigand 1994). These soils typically support mountain big sagebrush steppe communities. Mean fire intervals between 12 and 25 years occurred in these shrub steppe communities (Houston 1973, Burkhardt and Tisdale 1976, Martin and Johnson 1979, Miller and Rose 1999). However, Juniper Mountain located east of Alkali Lake in south central Oregon is an exception (Fig.4). This site may serve as a model as to what most of the mountain big sagebrush type would have looked liked if fire had played a minor role in the sagebrush ecosystem. This is the only old-growth stand we have measured throughout the range of western juniper that exceeds 30% canopy cover. On the north and northeast aspects the tree canopy cover ranged between 35 and 50%. On south and southwest aspects tree cover ranged between 20 and 35%. Shrub cover accounted for less than 1% of the understory cover. Dominant herbaceous species were Idaho fescue on the north aspects and Thurber needlegrass on the south aspects.

Sedimentary soils in the John Day Ecological Province

Very little work has been conducted on old-growth juniper on these soils. The majority of these soils occupied by old-growth western juniper occur in the John Day province with limited amounts occurring in the High Desert and Klamath provinces (Fig 1). These soils usually support a very low density of trees and sparse understory incapable of carrying fire (Fig. 5). The accumulation of both down and standing dead and decadent trees on many of these sites indicates the presence of very old stands. Tree ages on these soils exceed 1,000 years. Dead trees may remain standing for hundreds of years. The center growth rings on several of these trees go back 100 BC. These old-growth stands probably account for less than 3% of the juniper woodland component in the John Day province.

Figure 4. Dense ancient western juniper woodland growing on the north aspect of Juniper Mountain, in eastern Oregon. Soils are deep loamy Argixerolls and Haploxerolls. Tree canopy cover varies from 35 to 50%.

Figure 5. Ancient western juniper trees growing on sedimentary soils in central Oregon.

Aeolian sands in the Mazama and western High Desert Ecological Provinces

The aeolian sand region is located in the Mazama and northwestern portion of the High Desert Ecological Provinces, just east of the Cascade Mountain range (Fig.1). This region supports the most extensive stands of old-growth western juniper woodlands (Fig. 6). These old woodlands probably account for over 10% of the juniper woodlands in the Mazama Province. Soils in the Mazama Province are strongly influenced by Mazama pumice. In the northwest corner of the High Desert Province, sources of wind blown sands are primarily from Pleistocene lake beds, and Mount Mazama and Newberry Craters pumice. Stand structure varies across these provinces but are generally open with tree canopy cover typically ranging between 10 and 20%. Live tree density ranged between 41 and 123 per acre, standing dead ≤9 per acre, and down dead ≤7 per acre. Very slow decomposition rates allow for an accumulation of dead wood on these sites. Burned stumps and standing weathered trees can persist for hundreds of years. Old-growth stands were typically mixed age with 63% of the trees > 200 years (Waichler 1998). Several trees have been aged between 1,200 and 1,600 years old. Fires were typically small, burning single to several trees within a stand. However, old fire scars on these landscapes indicate occasional extensive fires burning in these stands. Idaho fescue, western needlegrass, and bluebunch wheatgrass (primarily on the west and southwest aspects) frequently dominate the understory. However, in the Bend and Redmond area which lies below 5,000 feet, rabbitbrush and cheatgrass will dominate the understory on sites that have been overgrazed or mechanically disturbed. In contrast to other provinces, tree removal by cutting also results in a loss of Idaho fescue and an increase in cheatgrass and rabbitbrush.

Figure 6. Old woodland growing on aeolian sands. Tree canopy cover is 15% and dominant understory grass is Idaho fescue.

WILDLIFE VALUES

It is important that future wildlife work describes both woodland structure and stand age. Old-growth woodlands are typically more structurally complex than postsettlement woodlands. More than 80 species of animals use living trees with decay, hollow trees, snags and logs in the interior Columbia River Basin (Bull et al. 1997). Although their report excluded juniper species, current breeding bird surveys showed old-growth western juniper woodlands provide important habitat for many bird species. Preliminary results from our songbird surveys indicate an increase in cavity nesters in old-growth compared to postsettlement woodlands. Densities of cavity nesting mountain blue birds, red- and white-breasted nuthatches have been consistently greater in old stands. Some of our highest counts of mountain blue bird also occurred in shrub steppe communities adjacent to old-growth stands. Our lowest counts have been recorded in closed postsettlement stands. At this time we are currently collecting information on cavity densities and minimum tree ages where cavities are typically found. Wood rats also commonly nest in the hollow trunks of western juniper. In addition to wood rats, the abundance and diversity of small mammals is typically greater than in postsettlement woodlands (Willis and Miller 1999).

During the winter, a large abundance of frugivores, including western and mountain bluebirds, cedar waxwings, American robins, and townsend solitares have been reported in the extensive juniper stands in central Oregon in the Mazama Province (Contreas 1997). These stands are predominately open old-growth woodlands, with 15% or less canopy cover supporting good crops of juniper berries. Dense woodlands produce very few berries (Miller and Rose 1995). We have observed heavy berry crops on trees over 500 years old growing in relatively open stands. Tree density appears to have a greater effect on the potential berry production than tree age.

MANAGEMENT CONSIDERATIONS

Before we can address how we should manage old-growth juniper woodlands in the Intermountain West we must ask the question what should these old-growth stands be managed for? Old-growth juniper and pinyon woodlands make up a small percentage of the juniper and pinyon woodland. These old-growth stands are structurally and topographically more complex than the younger more abundant woodlands, adding biological diversity to the landscape and providing an important source of habitat for many organisms. Many of these stands are also very esthetically pleasing and provide recreational, cultural and spiritual opportunities. Kaufmann et al. (1992) states, "old-growth provides us with a tremendous opportunity for retaining or enhancing biological features unique to old-growth ecosystems." We should evaluate fire policies influencing these old stands including both fire suppression and let burn decisions. Recent changes in overstory and/or understory can alter the response of these communities to fire. However, continued fire suppression in some woodlands may increase the potential for large stand replacement fires. Fuel woodcutting also appears to be a rather wasteful use of this limited resource, unless cutting is designed to remove postsettlement trees and restore presettlement stand structure.

Studies are needed to determine and describe the range of old-growth characteristics throughout the Intermountain West. Mapping and inventorying old-growth woodlands is extremely important for developing management and land-use plans. Development of an old-growth woodland classification system used in inventories would prove helpful in developing management plans. We also need information on gap dynamics, tree mortality, and succession following disturbance. This information will allow us to predict how these woodlands respond to disturbance. It will also allow us to evaluate pre- and postsettlement changes in community structure and composition that have occurred in old-growth stands, define desired future conditions, and develop management programs to restore or maintain old woodlands. To be successful old-growth woodlands cannot be managed on a single tree basis but only at the community and landscape levels. These old stands are an important landscape component in the Intermountain West, because they support many plant and animal species and interact with adjacent plant community types.

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LONG TERM VEGETATION CHANGES ON THE "ISLAND" RESEARCH NATURAL AREA

M. Ann Fox and Lee E. Eddleman

SUMMARY

This study on the "Island" was undertaken primarily to determine vegetation change over a thirty year period. While only a few differences could be statistically tested, the major differences noted include 1) an increase in western juniper cover, 2) an increase in shrub cover, and 3) a more even mix of grass species rather than the dominance of bluebunch wheatgrass and cheatgrass as noted by Driscoll (1964a), and 4) a higher litter cover.

INTRODUCTION

The study of ecosystem dynamics and successional change requires the repeated measurement of plant communities. Opportunities to build on past data sets are few, especially in relatively undisturbed areas. The purpose of the study was to document the vegetation of the "Island", a near pristine Research Natural Area (RNA) in the central Oregon's *Juniperus occidentalis* Zone. The "Island" RNA is a walled 250 acre mesa, elevation 2400 ft, located within the Cove Palisades State Park between the confluence of the Deschutes and Crooked rivers, approximately four miles northwest of Culver, Oregon. Other than stories of two summers of use by sheep in the 1920s (Driscoll 1964a), mule deer and small herbivores have been the main grazers. Charcoal and strip areas dominated by grasses and sprouting shrubs bear evidence of small patch fires. In 1964 Round Butte Dam was built, and the Crooked and Deschutes river canyons were partially filled to form Lake Billy Chinook which now surrounds all but the southern end of the "Island".

Vegetation composition was first documented by Driscoll (1964a, 1964b). Our study compared present day vegetation cover data for shrubs, grasses and forbs in 1992 and 1993 to that of Driscoll in 1960 and 1961, some 30 years prior. We also examined western juniper (*Juniperus occidentalis*) cover over a 50 year time period using aerial photos of the "Island" RNA available for each decade back to the mid 1940s.

Climatic data records dating from 1949 through 1994 from the Metolius Weather Station, 6 miles NE, show an average annual precipitation of 10.5 in. with annual temperatures ranging from a mean max. of 62 °F to a mean min. of 31 °F.

Woody vegetation on the "Island" consists primarily of western juniper, Wyoming big sagebrush (*Artemisia tridentata ssp. wyominensis*), and antelope bitterbrush (*Purshia tridentata*). Major grasses include bluebunch wheatgrass (*Agropyron spicatum*), Idaho fescue (*Festuca idahoensis*), Thurber needlegrass (*Stipa thurberiana)*and Sandberg bluegrass (*Poa sandbergi*). Forbs are rather sparse. Plant associations described by Driscoll (1964a) include the major *Juniperus occidentalis/Artemesia tridentata/Agropyron spicatum* (*Juoc/Artr/Agsp*) association and the minor *Juniperus occidentalis/Purshia tridentata/Agropyron spicatum* (*Juoc/Putr/Agsp*) association.

METHODS

The macroplot size and layout described by Poulton & Tisdale (1961) and used by Driscoll (1964a) was used in this study. Raw data, field notes, and the exact locations of their macroplots which had been marked with metal stakes were unavailable (personal communication with Richard Driscoll, 1992 and Ed Dealy, field assistant, 1992). However the two plant associations were easily relocated. We used the same restrictions in plot location as in the original study (Driscoll, personal communication 1992). Our study measured vegetation in both plant associations during two summers, 1992 and 1993, using 14 macroplots.

The *Juoc/Artr/Agsp* macroplots were located throughout the central longitudinal area of the "Island" and in the southern two/thirds latitudinal area. The *Juoc/Putr/Agsp* association was located on 35 acres in the northeast section of the "Island" and four macroplots were set up within it. We were also interested in other smaller areas where bitterbrush was found. Driscoll (1964a) made no mention as to the presence of these communities. Three macroplots were established, one each in the southeast, southwest, and west central areas of the "Island" where bitterbrush also was present.

Percent foliage cover was estimated for all herbaceous species and percent surface cover was estimated for rock, mosses/lichens, bare ground, and litter on microplots. Shrub cover was measured by line intercept, and height and diameter of shrubs were measured on belt transects. Aerial photographs were used to determine canopy cover of western juniper trees at six time intervals over a fifty year period including 1944, 1951, 1961, 1975, 1985, and 1995. Enlarged greyscale photocopies were made of aerial photos, and a plastic line/dot grid placed over the sites to estimate percent cover of the larger and thus visible western juniper trees within each association (Moessner 1960, Driscoll 1964a).

Our results and those of Driscoll (1962, 1964a, 1964b) were compared for statistically significant differences using Student's t-test, but only for those categories and species for which Driscoll reported standard errors.

RESULTS

The 1949 through 1994 climatic data from the Metolius Station showed temperature patterns to be fairly consistent. However, precipitation patterns were very erratic over the 45 year period. The month of highest precipitation occurred in all months of the year. While there were small clusters of years with below average precipitation (10.5 in.) and other clusters with above average precipitation, the majority of those below were intermittent with normal years while those above showed no particular pattern. Period precipitation extremes varied from 17.5 in. during 1983 to 6.2 in. during 1985.

We used an October through September water year for comparison purposes which showed that precipitation in the year preceding and the two years during each study were 7.5, 7.1, and

11.8 in. respectively for the 1960s study and were slightly higher at 8.2, 9.1 and 14.1 in. respectively for the 1990s study. Seasonal precipitation patterns differed with more than two times as much in the spring and about three times as much in the summer during the1990s study as compared to the 1960s study (Table 1).

Table 1. Precipitation at the Metolius, OR Weather Station for the year preceding and the two years of each field study for Oct.-Sept. water years*(inches).

	1960s			1990s		
	1958-'59	1959-'60	1960-'61	1990-'91	1991-'82	1992-'93
Oct-Mar "recharge"	5.6	5.5	9.0	3.0	4.8	8.3
Apr-Jun 'spring"	1.2	1.1	2.4	3.7	2.9	4.4
Jul-Sep 'summer"	0.7	0.5	0.4	1.5	1.4	1.4
Oct-Sep "water Yr"	7.5	7.1	11.8	8.2	9.1	14.1

- data from U.S. Weather Bureau (1994)

Recorded differences in vegetation between the 1960s study and the 1990s study indicate vegetation change has occurred (Tables 2). Statistical comparisons were made only for the percent cover of big sagebrush, bluebunch wheatgrass, and Sandberg bluegrass in the *Juoc/Artr/Agsp* association.

Cover for bluebunch wheatgrass and Sandberg bluegrass had changed with the former having lower cover values and the latter having higher cover values in the 1990s study versus the 1960s study. No significant differences in cover were found for western juniper and big sagebrush in the *Juoc/Artr/Agsp* association.

Total shrub cover appears higher in the 1990s study compared with the 1960s for both the *Juoc/Artr/Agsp* and the *Juoc/Putr/Agsp* associations. The major shrub involved was big sagebrush, however grey and green rabbitbrush (*Chrysothamnus nauseosus*, C. *visidiflorus)* and grey horsebrush (*Tetradymia canescens)* also made contributions. Total grass cover appears to have changed little in the two associations. However in both associations the cover of bluebunch wheatgrass and cheatgrass (*Bromus tectorum)* appears to have declined. Conversely, Sandberg bluegrass, Idaho fescue, and Thurbers needlegrass cover appears to have increased, providing a fairly even representation of the four perennial grass species.

Perennial forb cover was very low in both studies for both plant associations. Driscoll (1964a) mentioned a paucity of forbs, which appeared to be even more pronounced in our study. Annual forb species were numerous, but their cover appeared to be low and their constancy was variable from year to year (Driscoll 1964a). In our study they were identified as present only because of their scarcity.

Driscoll (1964) found striking differences in bare ground and litter cover between the two associations, with more bare ground found in the *Juoc/Artr/Agsp* plots (41% vs 19%) and more litter in the *Juoc/Putr/Agsp* plots (62% vs 31%). Assuming that Driscoll included rock cover with bare ground and moss/lichen cover with litter, we found similar relationships but a less dramatic difference between the two associations than was recorded in the 60's study. In the 90's study bare ground cover was generally higher on the *Juoc/Artr/Agsp* plots (35%) than on the

Juoc/Putr/Agsp plots (22%) while litter cover was higher on the *Juoc/Putr/Agsp* plots (57%) than on the *Juoc/Artr/Agsp* plots (46%).

Table 2. Cover values (%) for plant species and groups from macroplots on the Island for 1960-61 and for 1992-93 from the *Juoc/Artr/Agsp* association and from the *Juoc/Putr/Agsp* association. * and ^ indicate differences are significant($p<0.05$).

	Juoc/Artr/Agsp		**Juoc/Putr/Agsp**	
Plant Species	**60's**	**90's**	**60's**	**90's**
Wyoming Big Sagebrush	8.5	15.4		2.1
Bitterbrush	0.0	0.0	8.7	9.8
Grey Rabbitbrush	1.1	tr.	0.6	0.3
Green Rabbitbrush	0.0	tr.		0.5
Grey Horsebrush	0.0	0.0		0.4
Total Shrubs	**9.6**	**15.4**	**9.3**	**13.0**
Bluebunch wheatgrass	9.2*	3.4*	6.2	2.5
Idaho fescue	0.4	2.6	0.5	4.8
Thurbers needlegrass	2.0	2.9	1.1	4.6
Junegrass				0.1
Squirreltail	0.1	tr.	tr.	0.2
Sandberg bluegrass	1.3*	2.8*	0.4	3.9
Cheatgrass	1.7	0.1	12.4	2.6
Sixweeks fescue	0.6	0.4	0.3	0.6
Total Grasses	**15.3**	**12.3**	**20.9**	**19.3**
Western yarrow	0.1	tr.	0.6	0.1
Mtn. Dandelions	0.1	0.0		
Milkvetch	0.3	0.1	0.1	tr.
Nineleaf lomatium	0.6	tr.	0.1	tr.
Douglas phlox	0.1	0.0	0.1	tr.
Foothill death camas	0.0	0.0	0.1	tr.
Total Forbs	**1.2***	**0.1***	**1.0^**	**0.1^**

DISCUSSION

Canopy cover (%) of western juniper on the "Island" was measured using aerial photographs taken in 1944, 1951, 1961, 1975, 1985, and 1995. Tree cover for the *Juoc/Artr/Agsp* sites were not different from that of the *Juoc/Putr/Agsp* sites in any of the years measured. Nor was the percent cover on the "Island" generally different from decade to decade, but there was significantly more cover in 1995 when compared to 1944. Trees were unevenly clustered on the "Island" resulting in high sample variance especially in the *Juoc/Artr/Agsp* association which covers 21 acres. In general, tree cover appears to have increased gradually but steadily from 1944 to 1995 in both associations. Because of the method of using aerial photographs to measure tree cover, only larger trees were visible, thus it was likely that percent cover was underestimated.

When combining all of the macroplots on the "Island" into one unit and comparing the vegetation generally with Driscoll's (1964a) combined macroplot data, there appeared to be an increase over the 32 years in tree cover from 5% to 9.7%. Total shrub cover appeared to be higher, grass cover appeared to be lower, and forbs, a very minor component, appeared to have decreased, bare ground/rock cover stayed approximately the same, and litter cover (which includes moss/lichen) appeared to have increased more than 30 % in the *Juoc/Artr/Agsp* association.

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WESTERN JUNIPER ENCROACHMENT INTO ASPEN STANDS IN THE NORTHWEST GREAT BASIN

Travis Wall, Rick Miller, and Tony Svejcar

SUMMARY

Western juniper (*Juniperus occidentalis*) is invading and replacing quaking aspen (*Populus tremuloides*) stands in the Northwest Great Basin. Peak establishment of western juniper into aspen began at the turn of the century. Seventy-seven percent of aspen stands sampled have either been replaced, are dominated, or actively establishing populations of western juniper. The primary reason for western juniper encroachment into aspen is the lack of fire since the turn of the century and heavy browsing by wild and/or domestic ungulates. As western juniper replace these aspen stands, percent bare ground increases, herbaceous plant cover decreases, and soil properties are altered.

INTRODUCTION

Quaking aspen communities are declining due to encroachment of western juniper in the Northwest Great Basin (northern Nevada, northeastern California, and eastern Oregon). Although these communities constitute a small portion of the landscape across the Great Basin they contribute significantly to the biodiversity of wildlife and plant species. In aspen/grass stands located in the Great Basin of southeastern Oregon, 84 wildlife species reproduce and 110 wildlife species forage within these sites. In aspen/mountain big sagebrush, 95 wildlife species reproduce and 117 wildlife species forage within these sites (Maser et al. 1984). From this information, one may conclude that only riparian areas exceed aspen sites for the greatest majority of wildlife use proportional to total land area in southern Oregon.

Aspen communities generally have more lush undergrowth than neighboring coniferous forests (Mueggler 1985a). Aspen undergrowth can vary from less than 500 lb/acre to over 4,000 lb/acre. This variability is due to different environmental conditions, levels of animal use, and successional status of individual stands (Houston 1954, cited by Mueggler 1985b). If conifers overtake an aspen site, the understory vegetation is altered (Mueggler 1985a). Conifers more effectively shade the forest floor and change herbaceous quantity and species composition. Undergrowth in aspen communities decreases as the number of conifers increase (Mueggler 1985b). Bartos and Campbell (1997) state that when conifers overtake aspen communities, less water is available to the watershed, understory biomass vegetation is significantly reduced, and the diversity of wildlife and plant species declines. The greatest concern over conifer invasion is the permanency of aspen exclusion from succession once a climax conifer community persists.

In the Rocky Mountain region, Utah alone has 1.6 million acres of aspen (Mueggler 1988). Utah's aspen dominated lands have decreased by approximately 60% since European settlement (Bartos and Campbell 1997, 1998). With this in mind, one might ask if aspen in the Northwest Great Basin have decreased since settlement? And, are losses similar to those reported for aspen in Utah?

Today, western juniper inhabits over 8 million acres in the Northwest Great Basin. Extensive study has shown that western juniper is expanding its range into meadows, shrub-grasslands, riparian areas, and aspen stands (Miller and Wigand 1994, Miller and Rose 1995, Miller 1996). The most susceptible aspen communities are below 7,000 ft in elevation because this marks the upper elevation limit for western juniper (*Juniperus occidentalis* spp. *occidentalis*). On Steens Mountain in eastern Oregon, Miller and Rose (1995), found the greatest densities of western juniper occurred in aspen stands verses sagebrush community types. In these locations, western juniper are invading and replacing aspen. Due to aspen's limited distribution and ecological importance in the Northwest Great Basin, western juniper encroachment is very alarming.

The magnitude of aspen loss and the extent and effects of western juniper invasion into aspen communities in the Northwest Great Basin are unknown.
Loss of aspen warrants attention because of the ecological diversity it adds to landscapes predominated by sagebrush and juniper. By gaining a better understanding of western juniper encroachment into aspen communities, land managers and owners can make effective and proper decisions on how to perpetuate and maintain the aspen community.

OBJECTIVES

The focus of this study was to determine the extent and ecological effects of western juniper encroachment into aspen stands in the Northwest Great Basin. The objectives of the study include:

1. Determine the extent of western juniper invasion into aspen stands in the Northwest Great Basin.
2. Assess aspen stand age or time since last disturbance and when western juniper began to significantly invade aspen stands.
3. Determine the effects of aspen stand structure (density, cover, and age) on western juniper encroachment.
4. Determine the effects of western juniper invasion on soils previously influenced by aspen. Soil characteristics studied included C, N, and pH.
5. Measure the difference of C and N in aspen and western juniper litter.
6. Determine presettlement disturbance intervals in aspen stands.

STUDY AREA

Aspen stands typically were located along the north and northeast base of ridges where wind deposition causes excess snow accumulation. Elevation of aspen stands varied between 4900 and 7000 ft. Geographic locations of aspen communities studied in the Northwest Great Basin included:

Southeast Oregon	**Northwest Nevada**	**Northeast California**
Steens Mountain	Sheldon Refuge	Cedar Creek
Lakeview region:	Massacre Rim	McDonald Peak
Abert Rim	Mosquito Lake	Nelson Corral
Fishcreek Rim		
Coleman Rim		
Long Canyon		
Chewaucan Drainage		

METHODS

A 50 ft circular plot was laid in the middle of each aspen site sampled. Within this plot age, density, and canopy cover was measured for both aspen and western juniper. Percent bare ground and herbaceous plant cover was also measured. Soils were collected from aspen stands with no juniper and from aspen stands that had been completely replaced by juniper. These soils were then sent to a soils lab to determine differences in soil characteristics such as plant available nutrients and pH. Aspen and western juniper litter was collected from litter traps within stands and analyzed for differences in C and N.

RESULTS AND DISCUSSION

Extent of Western Juniper Encroachment Into Aspen

Western juniper is encroaching into aspen stands throughout the entire study area (below 7,000 ft elevation). Of the 91 aspen stands sampled, 86 (or 95%) contained various densities of western juniper. The average density of western juniper is 637 trees per acre of aspen (Fig. 1). These densities include all size classes from large adults to small juveniles. Twelve percent of aspen stands sampled have been completely replaced by western juniper. Twenty-three percent of aspen stands are in a state of regression with well-established western juniper populations on the brink of overtaking them. In 42 % of aspen stands sampled, western juniper is very common but not yet dominating.

As old aspen die, understory western juniper trees replace the aspen filling in the canopy holes. This is partially due to limited aspen recruitment. Statistical canopy cover data from this study shows that as aspen canopy cover decreases, juniper canopy cover increases. Presently, average juniper canopy cover in aspen stands is 27%. Fifty-one percent of aspen stands have over 10% juniper canopy cover. In 21% of the stands, western juniper accounts for to over 40% canopy cover. As western juniper canopy cover increases, bare ground increases and herbaceous plant cover decreases.

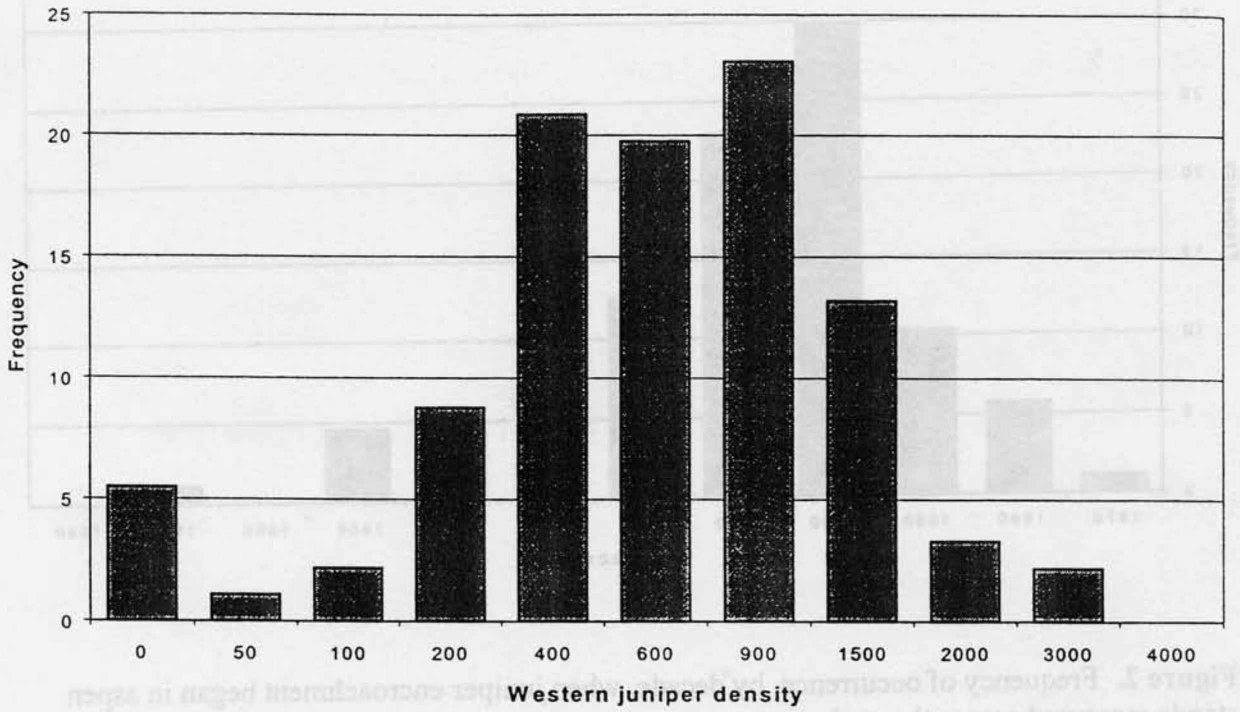

Figure1. Frequency of occurrence of different levels of western juniper densities across aspen stands sampled.

Succession of juniper into aspen communities has been attributed to events that occurred around the turn of the century. From aged tree data, western juniper encroachment peaked from 1900 to 1929 with 66% of the sampled trees establishing during this period (Figure 2). Only 5% of western juniper occurring in aspen stands are greater than 100 years old and no trees were older than 145 years. Reasons for expansion of young juniper include: optimum climatic conditions at the turn of the century, reduced fire resulting from European settlement with subsequent livestock use and fire suppression, and an increased seed source (Miller and Wigand 1994, Miller and Rose 1995, Miller 1996). Browsing by domestic and wild large herbivores has also limited aspen recruitment. Previous to this time, frequent fire events excluded juniper from aspen stands and perpetuated aspen stand vigor through suckering. Today, fire in aspen stands is considered an unusual event (DeByle et al. 1987). As a result, old and decadent stands continue to deteriorate throughout the West (Jones and DeByle 1985, DeByle et al. 1989). Thus, western juniper maintains an established foothold in the under-story, subsequently fills in canopy gaps as old, adult aspen die, and eventually replaces the entire stand.

Aspen Stand Dynamics

What is the current state of aspen stands in the Northwest Great Basin? Aspen stands sampled across the study area average 98 years old. Forty-seven percent of these stands are greater than 100 years old (Fig. 3). Since aspen longevity is relatively short, stands over 100 years old begin to regress. As individual aspen die, holes open in the canopy. In uneven-aged

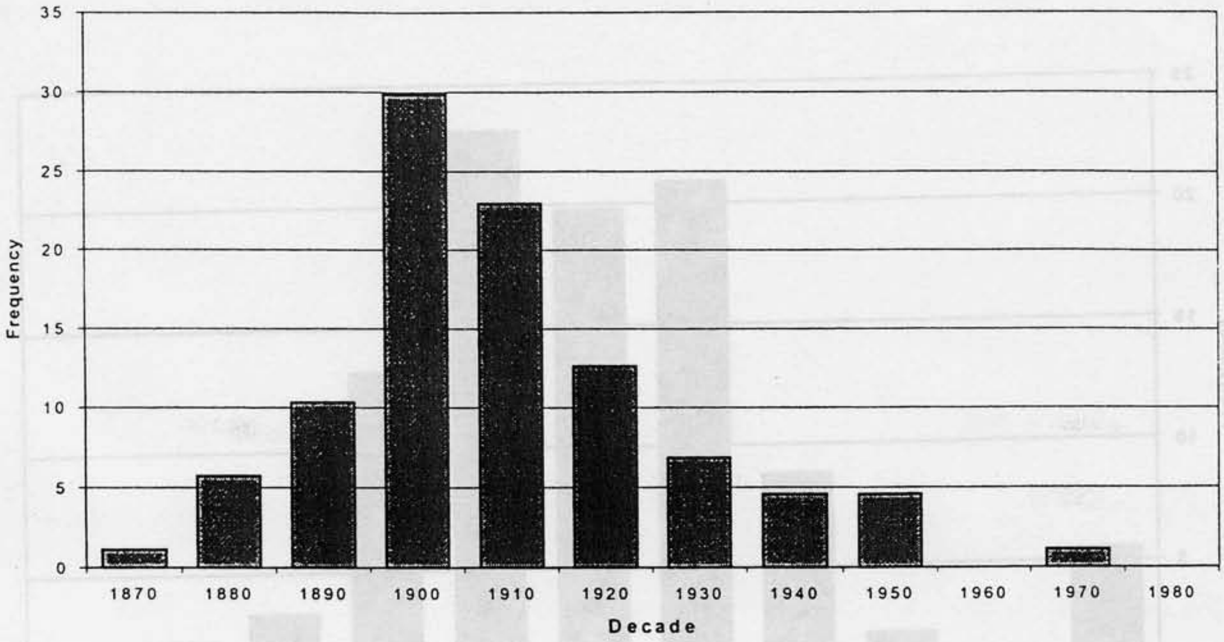

Figure 2. Frequency of occurrence, by decade, when juniper encroachment began in aspen stands measured across the study area.

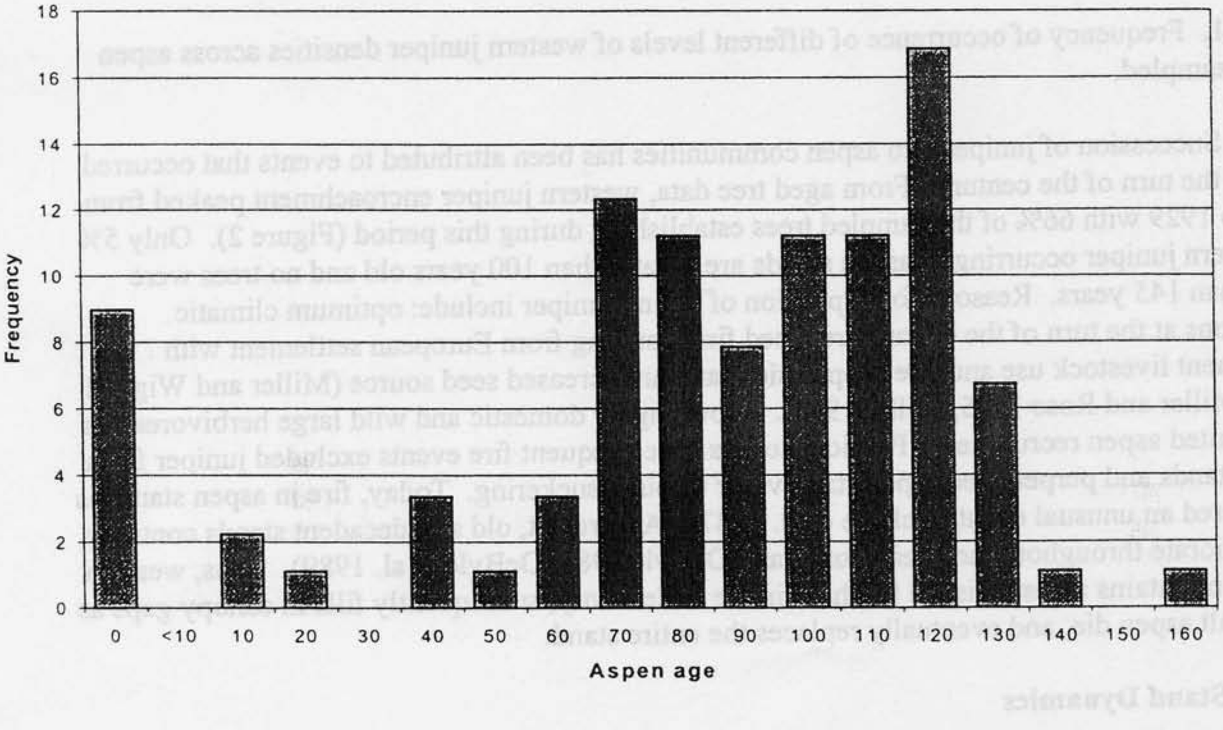

Figure 3. Frequency of occurrence aspen age for dominant canopy cover trees across the study area.

stands of aspen, younger aspen will fill these openings. However, most stands in the northwest Great Basin are even-aged. When an individual tree dies, no juvenile aspen are available to immediately fill the canopy gap. Of all aspen sites sampled across the study area, 70% had no active recruitment of new aspen. The other 30% had varying degrees of recruitment which still may not sustain aspen stands experiencing encroachment by longer-lived juniper. Additionally, aspen stands across the study area have an average of 20% dead trees in the stand. This perpetuates aspen stand decline because live adult aspen continue to release auxins that inhibit aspen regeneration while sun-loving juniper receive sufficient light in the under-story to grow and fill the gaps of dead and dying aspen.

Effects of Western Juniper Encroachment on Soil Previously Influenced by Aspen.

What changes, if any, does western juniper encroachment cause to soils? To find out, two categories of soils were sampled: 1) soils influenced by aspen, and 2) soils once influenced by aspen but now dominated and influenced by western juniper.

Soils influenced by western juniper had a higher C to N ratio and a higher pH than the aspen soil. By having a higher C to N ratio, less N is available for plant growth. This higher C to N ratio in soils can be attributed to western juniper litter as it also has a higher carbon to nitrogen ratio than aspen litter. Additionally, western juniper binds-up nutrients within the tree, not recycling it back to the soil as quickly as aspen since it is longer-lived and evergreen. Soils influenced by western juniper also had higher amounts of salts, lime, and sulfate, and lower amounts of magnesium, iron, manganese, and copper.

CONCLUSION

The majority of aspen stands sampled in the Northwest Great Basin were in various stages of western juniper encroachment. Seventy-seven percent of aspen have either been replaced, dominated, or rapidly establishing populations of western juniper. The primary reason for western juniper encroachment into aspen is lack of fire since the turn of the century and over browsing of young suckers by domestic and wild herbivores. Frequent fire events exclude juniper from aspen stands and induce suckering thus perpetuating the stand. However, in the absence of fire, western juniper will continue to replace aspen. In southern Colorado fire return intervals in aspen communities were estimated at 60 years (Romme et al. 1996). Long term effects of juniper to soil may render the site unsuitable for aspen and its associated plant component. To sustain or reclaim aspen, prescribed fire and allowed natural fire needs to occur at pre-settlement intervals. The majority of stands need an urgent proactive plan for fire reintroduction. Fire treatments should continue over space and time to invigorate the stands and deter future western juniper invasion. Overuse of young aspen suckers must also be decreased to allow for adequate aspen recruitment.

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FIRE HISTORY AND WESTERN JUNIPER ENCROACHMENT INTO SAGEBRUSH STEPPE[1]

Rick Miller and Jeff Rose

SUMMARY

The recent expansion of juniper into sagebrush steppe communities throughout the semiarid Intermountain West is most frequently attributed to the reduced role of fire, introduction and overstocking of domestic livestock in the late 1800s, and mild and wet climate conditions around the turn of the century. This statement, however, has limited factual support. Few studies of fire history in the sagebrush steppe have been published, and none that examine the timing of changes in mean fire intervals, introduction of livestock, and coincident climatic conditions with the initiation of postsettlement juniper expansion. This study was undertaken to determine the timing sequence of the postsettlement expansion of juniper, the introduction of domestic livestock, reduction in fire frequency, and optimal climate conditions. We documented the fire history and chronicled western juniper woodlands for a sagebrush steppe in a 5,000 ha watershed in south central Oregon. Records dating the introduction and buildup of livestock during the late 1800s and dates of initial fire suppression were summarized. Western juniper expansion began between 1875 and 1885, with peak expansion rates occurring between 1905 and 1925. The fire record spans 1601 to 1996. Before 1897, mean fire intervals within individual clusters ranged from 12 to 15 years with years between fires varying from 3 to 28. Nearly one-third of the fires in the basin were large and usually proceeded by one year of above-average tree-ring growth. Two fire events were recorded in the sparsely vegetated low sagebrush site, 1717 and 1855. The last large fire occurred in the study area in 1870, and the last small fire burned in 1897. The time sequence of wet climatic conditions between 1870 and 1915, introduction of livestock, and the reduced role of fire support the hypothesis that these factors contributed to the postsettlement expansion of western juniper.

INTRODUCTION

A statement we frequently take for granted is: " postsettlement expansion of juniper woodlands in the West is primarily attributed to the introduction and overstocking of livestock, the reduced role of fire, and optimal climatic conditions during the late 1800s". However, only a handful of studies have documented the mean fire intervals in the sagebrush steppe biome, and few, if any, have evaluated the chronosequence of the introduction of livestock, the reduced role of fire, and climatic conditions with the initiation of postsettlement woodland expansion.

Expansion of western juniper *(Juniperus occidentalis),* sagebrush steppe communities in the interior Northwest United States coincides with Euro-American settlement (Burkhardt and Tisdale 1976, Young and Evans 1981, Miller and Rose 1995). Pinyon-juniper woodlands have also expanded during this period in Utah and Nevada (Tausch and West 1988). The current expansion of woodlands is unparalleled to previous expansions, which occurred over the past

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[1] This paper is an abridged version from R.F. Miller, and J.A. Rose. 1999. Fire History and Western Juniper Encroachment in Sagebrush Steppe. Journal of Range Management. 59: (in press).

6,000 years (Miller and Wigand 1994). Presettlement expansions occurred during cool wet periods while the current expansion is occurring during a warmer drier period. Current tree densities are also considerably greater than past Holocene expansions based on juniper pollen abundance.

The recent expansion of western juniper began during the late 1800s (Young and Evans 1981, Eddleman 1987, Miller and Rose 1995). The majority of present day woodlands are less than 100 years old (USDI-BLM 1990). Relict *Juniperus* woodlands, historical documents and photographs, pollen and macro fossil data, and the absence of stumps or logs in mountain sagebrush communities indicate presettlement woodlands were considerably less abundant in the West. Old trees are primarily confined to rocky surfaces or ridges with sparse vegetation (West 1984, Miller and Wigand 1994, Miller and Rose 1995). Trees in these locations have the ability to exceed ages of 1,000 years (Miller et al. 1999).

Fire is thought to have played an important role in shaping sagebrush steppe communities and limiting the expansion of juniper in the Intermountain West before Eurasian settlement. In the semiarid region of the Intermountain Northwest, presettlement mean fire intervals between 15 to 25 years have been reported for the mountain big sagebrush community type (Houston 1973, Burkhardt and Tisdale 1976, Martin and Johnson 1979). However, during the last 100 years, the role of fire has greatly declined in these ecosystems. The decline in fire has been attributed to reduced fine fuels through heavy livestock grazing in the late 1800s and to reduced anthropogenic set fires during the nineteenth century (Burkhardt and Tisdale 1976, Miller et al. 1994).

Optimal climatic conditions during the late 1800s and early 1900s may have also interacted with the reduced role of fire and overgrazing by domestic herbivores to accelerate the rate of western juniper expansion into shrub-steppe communities. During this period, winters became more mild, and precipitation increased above the current long-term average conditions (Antevs 1938, Graumlich 1987), which promotes vigorous growth in western juniper.

This study was designed to 1) chronicle western juniper establishment, 2) document pre- and postsettlement mean fire intervals in a mountain big sagebrush community type and 3) determine the proportion of large-to-small fires and evaluate their relationship to growing conditions in years preceding fires. We hypothesized that postsettlement expansion of western juniper woodland was synchronous with the introduction and overstocking of domestic livestock, changes in mean fire intervals, and optimal climate conditions for plant growth.

METHODS AND STUDY AREA

Study Area

The study area was located within the Fremont National Forest in the upper Chewaucan River Basin, 8 km south of Paisley, Oregon. The study unit encompassed 12,000 acres. Vegetation on moderate to moderately deep soils was characterized by mountain big sagebrush with Idaho fescue dominating the north aspects and Thurber needlegrass dominating the south aspects. The low sagebrush/Sandberg bluegrass community type occupied the shallow heavy clay soils. Associated with these plant communities were juniper trees in varying levels of density. The long-term average precipitation was approximately 15 inches.

Introduction of livestock occurred in the late 1860s in the Chewaucan River Basin (Oliphant 1968). By November of 1873, approximately 4,000 cattle were reported in the lower river basin with several thousand sheep moving in the following year. During the next five years, livestock numbers increased rapidly and peaked at the end of the nineteenth-century. Since 1915 sheep have declined on the forest from nearly 400,000 AUM's (animal unit months) to less than 1,000, while cattle numbers have declined about 30 percent. The USDA Forest Service office in Paisley was established in 1908, which marked the beginning of fire suppression in the study area.

Plant community measurements

Two hundred and fifty points were randomly selected and surveyed in the study area. The major community type (dominant shrub and perennial grass), percent slope, aspect, elevation, stage of transition (Table 1, see page 14), presence or absence of presettlement juniper, old stumps, and logs were recorded. Thirty-two plots were chosen for intensive measurement through a selected random sample to represent the two major community types, low sagebrush and mountain big sagebrush, in various stages of woodland transition.

In 32-circular plots, the juniper density, height, and canopy cover were measured with a radius of 66 ft in mountain big sagebrush and 99 ft in low sagebrush communities. All juniper trees were counted and recorded in 1m height classes. Presettlement trees were also recorded. Tree canopy cover was estimated and all trees were cored or cross-sectioned within each plot to determine aging. The number of standing dead, stumps, and logs were also recorded for each plot.

Fire History

A limited number of small clusters of presettlement ponderosa pine trees were scattered across the study area. Four of these sites contained fire-scarred ponderosa pine trees. Fire history was documented by collecting partial cross sections from three scarred trees within three sites: I, III, and IV, and one cross section from site II. Fire-scarred ponderosa pine trees ranged in age from 275 to 590 years. Cross sections were cross-dated to assign accurate dates to each fire occurrence. Seasonality of fires was estimated from the relative position of the fire scar within the annual ring. We also attempted to determine the occurrence of fire events in the fuel limited low sagebrush / Sandberg bluegrass community. Datable cross-sections from twelve burned western juniper stumps and logs were collected in a general search across the Ennis Butte basin. Tree-ring indices 2 years before and during the year of fire occurrence were compared to evaluate growing conditions preceding and during the year of fire.

Definitions (derived from W. Romme 1980)

Mean fire interval - average time between fires for a designated area during a designated time period; the size of the area and the time period must be specified.

Fire occurrence - one fire event within a designated area during a designated time.

Fire frequency - the number of fires per unit of time in a designated area; the size of the area must be specified.

Fire chronology - a chronological listing of the total fires documented for a designated area, the dates being corrected by cross-dating.

Cross-dating - correcting the chronology determined from an individual tree-ring sample by comparing it with a master tree-ring chronology developed for the area.

RESULTS

Juniper Expansion

Western juniper began expanding between 1875 and 1885 in the Chewaucan River Basin. Tree establishment increased rapidly during the following decades. In the mountain big sagebrush community type, rate of expansion peaked between 1905 and 1915 with generally constant establishment in the open stands through 1995, with the exception of 1935-1945 (Fig. 1). The decline in tree establishment during this period may have been the result of the severe drought conditions during the 1930s. Mean age of trees across the study area was 68 years. Presettlement trees (>130 years) across the study area accounted for less than 1 percent of the total population. The absence of old stumps, logs, and charcoal suggested that presettlement trees have not occupied this community type in the recent past. Woody material may persist in this environment for over 130 years. In the low sagebrush community type, approximately 1.5 percent of the juniper trees measured were greater than 130 years old with several trees sampled exceeding 500 years.

Tree density and canopy cover in closed stands were 4-times and 8-times greater, respectively, than the open juniper shrub steppe stands (Table 2). Saplings were fairly common in open stands but absent in closed stands. Expansion of juniper in open and closed stands across community types began during the same period. However, age-class distributions were different between the closed stands in the mountain big sagebrush types and the open mountain big sagebrush and low sagebrush community types (Fig. 2). Juniper establishment in the closed stands peaked during 1905 to 1915, with 78 percent of the trees establishing between 1885 and 1925 (Fig. 2).

Fire History

Age of sampled, fire-scarred ponderosa pine trees ranged between 275 to 590 years. The fire record we analyzed spanned the period from 1601 to 1996 (Fig. 3). All scars occurred near or at the termination of tree-ring width development, indicating late summer and fall fires. Season of fire events was evenly proportioned during late summer and fall. Before 1871, a fire event occurred somewhere in the study area on an average of every 7.7 years. The number of years between two fire events occurring somewhere in the study area varied between 1 and 19. The maximum hazard function was 45 years (100 percent probability that a fire would occur

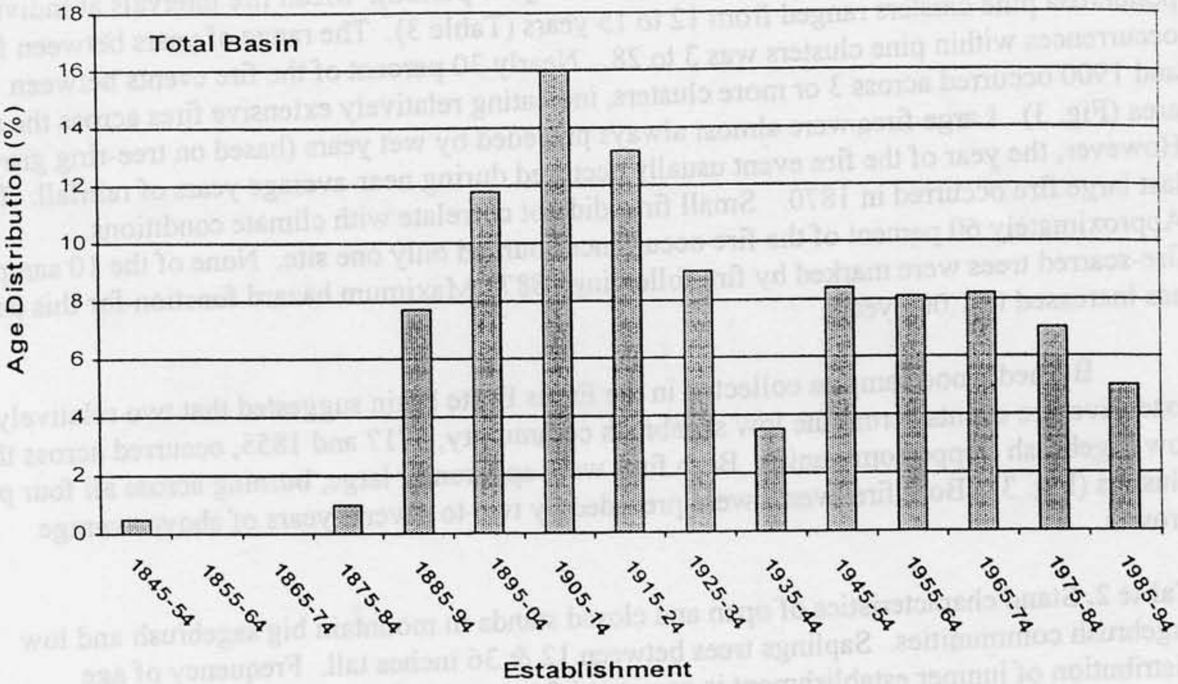

Figure 1. Age structure of western juniper trees by decade across the 32 plots (n=510) within the Chewaucan River basin.

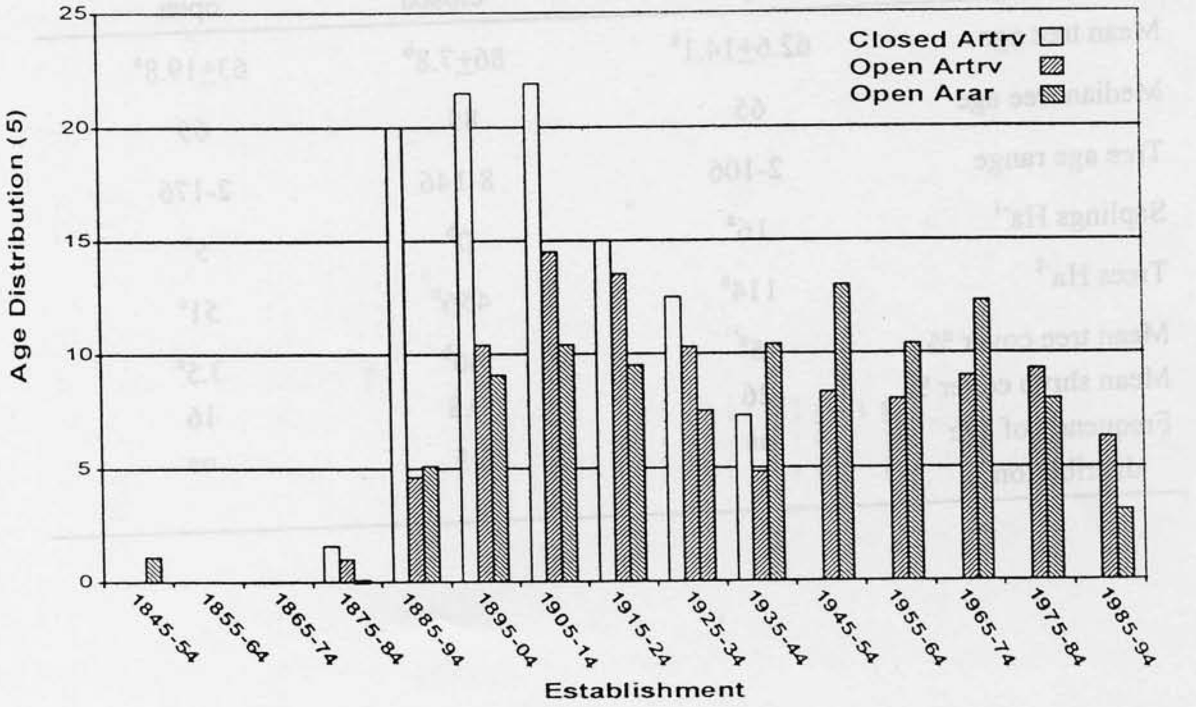

Figure 2. Age distribution (%) of western juniper trees by decade for open low sagebrush (ARAR) (n=88), and open and closed mountain big sagebrush (ARTRV) (n=422).

somewhere inside the study area in less than a 45-year period). Mean fire intervals at individual ponderosa pine clusters ranged from 12 to 15 years (Table 3). The range of years between fire occurrences within pine clusters was 3 to 28. Nearly 30 percent of the fire events between 1693 and 1900 occurred across 3 or more clusters, indicating relatively extensive fires across the study area (Fig. 3). Large fires were almost always preceded by wet years (based on tree-ring growth). However, the year of the fire event usually occurred during near-average years of rainfall. The last large fire occurred in 1870. Small fires did not correlate with climate conditions. Approximately 60 percent of the fire occurrences burned only one site. None of the 10 sampled, fire-scarred trees were marked by fire following 1887. Maximum hazard function for this period has increased to 1,000 years.

Burned wood samples collected in the Ennis Butte basin suggested that two relatively extensive fire events across the low sagebrush community, 1717 and 1855, occurred across the low sagebrush steppe community. Both fires were apparently large, burning across all four pine-clusters (Fig. 3). Both fire events were preceded by two to several years of above average growth.

Table 2. Stand characteristics of open and closed stands in mountain big sagebrush and low sagebrush communities. Saplings trees between 12 & 36 inches tall. Frequency of age distribution of juniper establishment is computed for decades from 1885 through 1995. Rows followed by a different letter are significantly different at the P=0.05.

	Mountain big sagebrush		Low sagebrush
	open	closed	open
Mean tree age	62.6±14.1^{a}	86±7.8^{b}	63±19.8^{a}
Median tree age	65	85	65
Tree age range	2-106	8-146	2-176
Saplings Ha^{-1}	16^{a}	0^{b}	5^{c}
Trees Ha^{-1}	114^{a}	456^{b}	51^{c}
Mean tree cover %	5^{a}	40^{b}	3.5^{a}
Mean shrub cover %	26	0.8	16
Frequency of age distribution	ns	s	ns

Table 3. Presettlement mean fire interval computed for the fire period in each cluster where sample size is ≥ 2 (except for site II, n=1). The minimum and maximum number of years between two fire events within a cluster are presented as the fire interval range. The mean fire interval and range for the study area indicates a fire occurred somewhere in the study area at a minimum mean fire interval of 7.7 years with years between fires varying between 1 and 19.

Site and Fire Period	Fire Interval	
	Mean	Range
I 1783-1870	12	3-23
II 1654-1870	27	12-54
III 1693-1870	12	4-28
IV 1601-1870	15	6-28
Study Area 1654 - 1870	7.7	1-19

CONCLUSIONS

Western juniper chronicled in the Chewaucan River basin clearly showed major structural change across sagebrush steppe communities since 1875. The most rapid period of establishment in mountain big sagebrush steppe occurred between 1885 and 1925, a period of wetter than average conditions, few fires, and intensive livestock grazing. The start of woodland expansion and its pattern of establishment do not support the hypothesis that rising levels of CO_2 was a primary factor driving 20th century succession of juniper woodland in the northern portion of the Great Basin. Before1880, fire probably played a major role in limiting western juniper encroachment into these sagebrush communities. Mean fire intervals of less than 15 years were adequate to inhibit western juniper encroachment and probably limit sagebrush cover, allowing the herbaceous layer to dominate the landscape. The co-occurrence of wet climatic conditions, introduction of livestock, and the reduced role of fire do support the hypothesis that all of these factors contributed to the postsettlement expansion of juniper in the West.

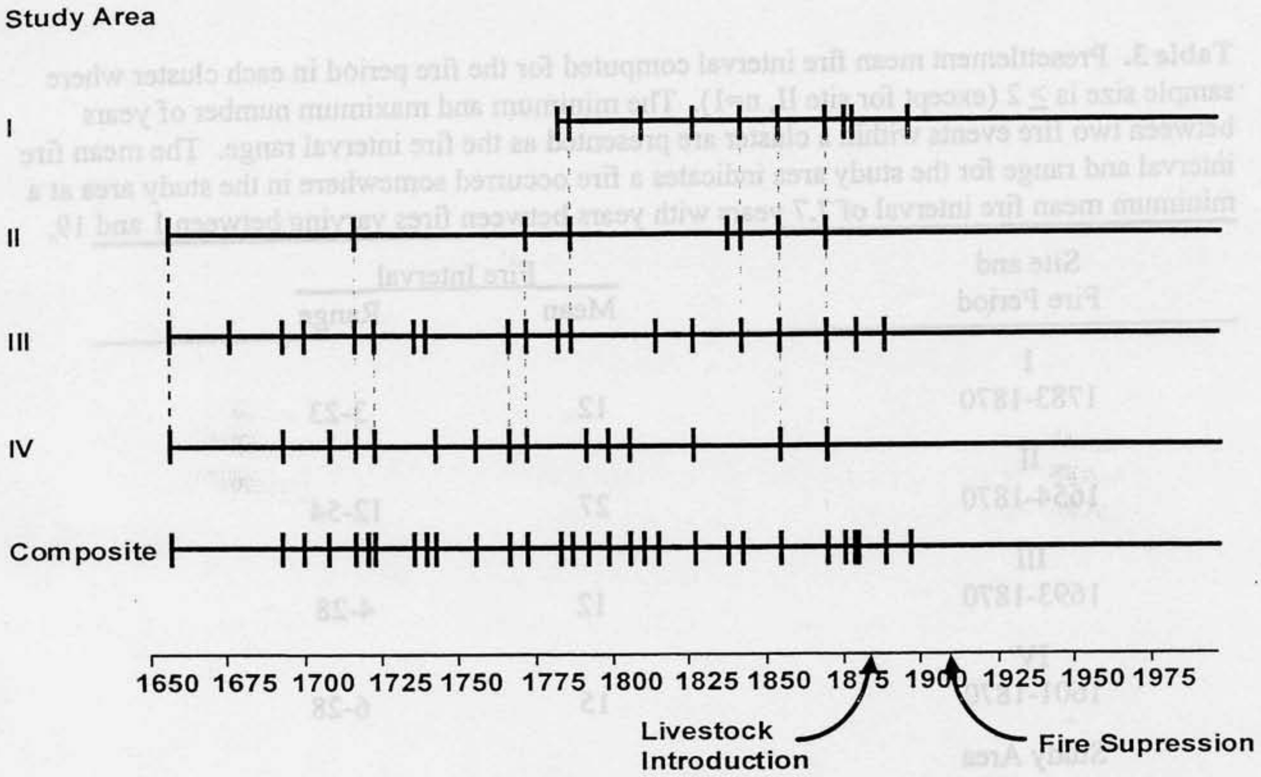

Figure 3. Master fire chronology for the mountain big sagebrush steppe community in the upper Chewaucan River basin. Fire history extends from 1601 to 1996. Each horizontal line represents a sample composite for each collection site with the bottom line being a composite for all fire scar samples across the four sites. Each vertical line designates a fire occurrence. Dashed lines connect collection sites where fires occurred across two or more sites in the same year.

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MANAGING WESTERN JUNIPER FOR WILDLIFE[1]

Rick Miller

SUMMARY

As woodland succession advances towards mature juniper woodlands, wildlife habitat changes for a wide array of species. It is important to identify both the type of site and stage of woodland development when considering management strategies and evaluating wildlife habitat values. If woodlands are allowed to fully develop plant diversity, structural diversity, berry production, and the opportunity to burn greatly decrease. Maintaining a balance between juniper trees with other plant forms such as shrubs, grasses, and wildflowers will provide the greatest opportunity for the maximum number of wildlife species at the community level. At the landscape level the greatest diversity of wildlife species will occur where there is a mosaic of communities in various stages of succession.

INTRODUCTION

Throughout the West, one of the most pronounced plant community changes in the 20th century has been the expansion of juniper and pinyon-juniper woodlands. These arid woodlands occupy over 42 million acres in the western United States. Western juniper (*Juniperus occidentalis* ssp. *occidentalis* Hook.) occupies approximately 5 million acres in eastern Oregon, 3 million acres in northeastern California, and approximately ½ million acres in northwestern Nevada and southwestern Idaho. Its northern most distribution is represented by a few stands in southeastern Washington.

Western juniper has rapidly expanded into neighboring plant communities during the past 120 years. As woodland succession advances towards mature juniper woodlands, wildlife habitat changes for a wide array of species. It is important to identify both the type of site and stage of woodland development when considering management strategies and evaluating wildlife habitat values.

JUNIPER ECOLOGY

In the Oregon high desert and south to Susanville, California, juniper trees older than 130 years are typically found on rocky surfaces and ridges where fuels from understory vegetation are limiting. These stands are typically open with widely scattered trees. In the pumice sand region just east of the Cascades, ancient woodlands characterize portions of the landscape. These intermittent old growth woodlands are distributed from the Connley Hills near Silver Lake north to the Bend area. The majority of these old stands are relatively open with tree canopy cover typically ranging between 5 and 15%. Old growth juniper trees can exceed 1,000 years of age and remain standing for hundreds of years after death. However, an estimated 97% of western juniper trees in

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eastern Oregon have established since the 1870s. Excessive grazing, reduced role of fire, and the alteration of plant community structure (the increase in shrub cover provides ideal sites for juniper establishment) in the late 1800s were the predominant factors initiating the rapid expansion of western juniper in the Pacific Northwest. The expansion of western juniper was further accelerated by wetter than average conditions between 1870 and 1915. The reduced role of fire and the increase in shrubs have allowed western juniper to readily encroach into many plant communities throughout its range. Plant communities most affected are mountain big sagebrush-bunchgrass, low sagebrush, aspen groves, and mountain mahogany located below 7,000 ft. Juniper is also expanding into riparian communities.

Female flowers are pollinated early in the spring by male cones developed during the previous growing season. Female cones (berries) grow to full size the first year, reach maturity the second year, and fall off during the second autumn and winter. Seeds are primarily dispersed by birds. Seeds are also spread across the landscape by rabbits, coyotes, and water flowing across the land surface. Trees typically reach full reproductive potential between 50 and 70 years of age and remain productive at very old ages. Seeds may remain dormant, germinating over a period of several years as cool moist conditions occur. The majority of successful seedling establishment occurs beneath sagebrush and other shrubs, which serve as nurse plants.

During the early stages of woodland development, juniper adds structural diversity to shrub steppe community types. As woodlands mature, the shrub layer dies out. Depending on the soils, grasses and wildflowers may or may not be lost in the understory. A good cover of grasses usually persists beneath a fully developed woodland on deep well-drained soils. This most often occurs on sites where Idaho fescue dominates the understory. However, the grass and wildflower layer is usually lost on soils with cemented hardpans at 24 inches or less in depth. This frequently occurs on south slopes occupied by Thurber needlegrass. Accelerated soil erosion can become a severe problem on these sites during the late stages of woodland development. Losses in mountain mahogany, bitterbrush, and aspen also occur as juniper gains dominance in these communities.

Although plant species and structural diversity increase during the early stages of juniper encroachment, diversity declines during the late stages of woodland development. The greatest structural diversity typically occurs during the mid phase of woodland succession where the shrub steppe component still remains intact. The successional stage of juniper woodlands in a mountain big sagebrush community type can be identified using the characteristics described in Table 1 (see page 14). Significant reductions in the shrub layer begin when the tree canopy reaches about 1/3 of full potential for the site. Juniper cover in closed woodlands (where juniper cover and density have reached full potential for the site) typically ranges from 25% in south aspect mountain big sagebrush communities to over 60% on north slopes with deep soils. Maximum juniper canopy cover in aspen communities approaches 90%.

WILDLIFE AND JUNIPER

Limited work in juniper woodlands has been conducted evaluating the relationship of wildlife species, particularly non-game species. In addition, much of the work did not quantify woodland composition and structure and ignored the stage of woodland succession. Over 100 wildlife species have been reported to use open juniper woodlands in eastern Oregon for thermal and hiding cover, nesting, and food. However, the amount of tree canopy and composition of the understory greatly affects the suitability of habitat for wildlife and thus influences species composition and abundance. Wildlife diversity in juniper communities is strongly related to the diversity and abundance of understory plant species. Animals often have a narrower adaptation to plant structure (e.g. the number of layers of vegetation present in a community such as trees, shrubs, tall grasses, short grasses, and mosses) than plant food species. For example, vertical layers of vegetation have been closely related to avian diversity. Open juniper woodlands have the greatest potential for maximum structural diversity with all layers present. However, as the tree canopy closes and juniper dominates the site the shrub layer is lost. Depending on the site and soils, the grass and wildflower layer may also be lost or greatly reduced. Juniper berry production, an important mast crop for many wildlife species, is also significantly reduced as woodland succession proceeds to stand closure. Open woodlands of today, which provide important habitat for many wildlife species, become less diverse as woodland succession continues to stand closure. Resource managers need to be aware of the dynamics of juniper woodland communities and develop long range plans to meet the desired objectives on a landscape level.

Large Herbivores

The amount of juniper cover, tree canopy depth, tree height, and stem density have been found to moderate severe winter conditions. In south central Oregon, woodlands that had 30% tree canopy, tree heights 15 ft tall, and 13 mature trees per acre reduced temperature severity, wind, and snow cover during the winter. Deer were observed to occupy these woodlands sites during severe winter conditions although food resources were limited. Survival rates of deer fawns were also greater during severe winter conditions in these woodlands that provided adequate thermal cover. However, woodlands that provide thermal cover typically offer little food for wildlife, or shrub cover for smaller species and poor cover for fawning. In eastern Oregon, optimal deer fawning habitat is characterized by a dense shrub layer (near 40%), typically found on productive north slopes with less than 5% juniper cover. Food plants are also generally more nutritious growing in open sunlight than under shady conditions created by dense tree canopies.

Little work has been done evaluating the relationship of other large wild herbivores with juniper. Antelope have been observed passing through open stands of juniper but typically prefer open shrub grassland communities. Their sense of security generally decreases as canopy height and density increases. Elk have been also observed in open woodlands. However, treatments that reduce conifer canopy densities and cover typically increase elk densities. Big horn sheep also generally prefer open habitats and are not a juniper obligate. They have, however, been observed to use juniper for shade during hot summer days on Hart Mountain.

Birds

Although a wide variety of birds utilize juniper communities, limited work has been done relating the successional stage or density of trees to bird diversity and abundance. Open juniper woodlands with understory diversity provide good habitat for ground, shrub, and tree nesting birds. However, in closed juniper woodlands, ground and shrub nesting birds are generally absent or present in very low numbers. The greatest avian use in closed woodlands generally occurs at the edge where the woodland adjoins other plant communities. Avian diversity and abundance can be very high in healthy shrub communities lacking trees. However, as trees establish in shrub steppe communities new species such as mountain chickadees, Oregon juncos, and chipping sparrows begin using these communities. The greatest avian diversity in the semiarid uplands typically occurs in these open juniper communities where the shrub, grass, and forb understory are still intact. As tree dominance increases and the shrub layer declines, populations of such birds as sage grouse, green-tailed towhee, western meadow lark, sage thrasher, and vesper, Brewers, and sage sparrows decline.

Juniper berries provide important food for a number of bird species, especially during the winter. Juniper berries are readily consumed by Townsend solitaires, American robins, mountain blue birds, cedar waxwings, Steller's jays, and scrub jays. Ripe juniper berries are an important source of energy containing 46% carbohydrate and 16% fat. During the winter, solitaries and robins consume over 200 berries per day.

Trees less than 100 years old rarely provide cavities. Old growth juniper stands can provide a good source of cavities for nesting birds. Cavities are most often observed in trees over 350 years old. Snags also provide a good source of cavities. Snag densities are usually low in old growth stands and absent in young stands because of very low tree mortality. Snag density in old growth stands in the pumice region is highly variable, typically averaging less than 3 per acre for trees greater than 14 inches in diameter. Dead trees can remain standing for up to 600 years.

Small Mammals

Both diversity and abundance of small mammals are closely correlated with the composition and structure of the tree understory. However, several species such as the pinyon mouse and woodrat are closely tied to juniper. Berries also provide a food source for some small mammals such as rabbits. Increases in small mammal populations, including the desert cottontail, have been reported following the cutting of juniper trees where the slash was left on the ground. The response of small mammal abundance and diversity is also closely linked to the response of shrubs, grasses, and wildflowers following juniper thinning or clearing. Seed production of understory plants usually increases following the reduction of the tree overstory, providing food for both small mammals and birds. As small mammal abundance increases so does food opportunities for raptors. Downed logs, which provide additional structure used by small mammals are uncommon in undisturbed juniper stands less than 130 years old and typically occur in very low densities in old growth stands.

Amphibians and Reptiles

We probably know the least about amphibian and reptile use in juniper woodlands. Fifteen reptiles and two amphibians species have been reported in communities containing western juniper. However, the presence of rock outcrops, ant mounds, and other site characteristics appear to have a stronger influence on the abundance and diversity of reptiles and amphibians than the presence or absence of juniper. In old growth stands we have observed a high frequency of use by the western fence lizard of juniper logs.

MANAGEMENT

The key to managing juniper for wildlife is not to totally remove it but to maintain a balance of juniper trees with other plant forms such as shrubs, grasses, and wildflowers. Two commonly used tools for managing juniper communities are fire and cutting. In the past, chaining was also a common practice for killing juniper trees but is now cost prohibitive. This technique requires two D-8 caterpillar tractors and 200 to 500 ft of ship anchor chain. The Division of Wildlife in Utah still uses a combination of chaining and seeding in pinyon-juniper woodlands to enhance big game habitat on their lands. Chemical control of western juniper has generally had mixed results and has been little used over the years. The best management tool or tools used for juniper woodland management depends on: (1) site potential and soils, (2) condition of the site, (3) stage of woodland development, and (4) objectives.

Managing for Structural Diversity

Managing juniper woodlands for wildlife should be considered at both the community and landscape levels. Landscapes, composed of a mosaic of plant communities, may be defined at different scales, including regional, a specific watershed, or a portion of a watershed. Maximum wildlife diversity at the community level will occur in juniper woodlands maintained in an open state with good shrub and herb cover. This can be accomplished by maintaining 5 to 12 full size trees per acre or tree canopy cover at less than 10%. On less productive sites, such as south slopes, less than 5% tree canopy cover should allow for vigorous understory growth. Thinning juniper to these levels can enhance bitterbrush leader growth two to four fold compared to growth under nearly closed juniper woodlands. When thinning a stand, old growth trees should be left. Open stands, where trees are not in competition with one another, will increase the potential for maximum berry crops, although such production is highly weather dependent.

At the landscape level, the composition, size, and shape of juniper woodlands and other plant communities greatly influence many wildlife species. How juniper woodlands fit into this mosaic of surrounding communities should be considered when developing management plans. For example, in one of our study sites in south central Oregon mountain bluebird densities were greatest where shrub communities were adjacent to old growth juniper woodlands.

Juniper Management on Good Condition Sites

Managing sites with an intact understory of native plant species provides a great opportunity for manipulating plant community structure and composition for optimal wildlife use. Fuel levels are typically sufficient on these sites, with the exception of low sagebrush communities, to allow the use of fire. Plant communities with a good composition of perennial native plants typically respond very well after fire. Grasses may or may not increase immediately following fire depending on the species present. Perennial forbs typically increase two to four fold. Native annual forbs also usually increase after fire. However, perennial forbs and shrub species with growing points above the ground will be significantly reduced by fire. Examples of fire sensitive species are mat forming forbs such as phlox, buckwheat, and sandwort, and shrubs such as sagebrush, bitterbrush, and mountain mahogany. In the absence of fire these shrubs will slowly decline as juniper woodland development approaches stand closure. At the landscape level, a mosaic of burn and unburned patches increases the overall diversity at this larger scale. Abundance and diversity of wildlife species is often similar between adjacent burned and unburned plant communities. However, wildlife species composition is usually different between burned and unburned communities, increasing the overall diversity of wildlife at the landscape level.

Fire prescriptions can determine the intensity of the fire, which will in turn influence the amount of area left unburned. There is more opportunity to create a mosaic under prescribed fire than under wildfire conditions. Juniper trees 10 ft tall or less are most susceptible to fire. However, the composition and structure of fuels and severity of environmental conditions at the time of the burn will determine tree mortality. The primary advantage of fire is that it can be an economical way of treating large areas. It is also a natural process that occurred prior to Eurasian settlement across many of these landscapes. The greatest disadvantage of using fire, particularly for private landowners, is liability. In addition, burning small units less than several hundred acres is both expensive and places the communities at risk from over grazing by domestic and wild herbivores. Cutting is a useful tool when treating smaller land areas, especially where desirable fire sensitive species such as bitterbrush are present in the understory. Cutting is selective, allowing the landowner to thin to the desired tree density and leave trees such as old growth or female trees for berry production. Liability is also low, but cutting is expensive and not practical for treating large areas.

Juniper Management on Poor Condition Sites

Communities in poor condition generally should not be burned. Fire typically can cause further degradation of these sites through increasing the dominance of exotic plants and bare ground. In addition, fuel is often not sufficient to use fire as a tool. However, under extreme climatic conditions crown fires can sweep across these communities especially if cheatgrass and/or medusahead are abundant in the understory. These exotic species will quickly reestablish following fire. As a rule of thumb invasion of exotic species is a greater threat below 5,000 ft elevation; above 5,000 ft native species are generally more competitive than these introduced Mediterranean annuals.

Cutting juniper can be used to help restore the condition of sites that have lost the majority of the understory cover and have large bareground interspaces between the trees, which make them highly susceptible to erosion. Cutting the trees and spreading the limbs across the surface creates ground cover that protects the soils from erosion, provides sites for seedling establishment of understory plants, and furnishes cover for wildlife. A minimum of two desirable perennial grasses or forbs per 10 ft^2 are necessary for restoring the understory on sites receiving less than 12 inches of annual precipitation. One plant per 10 ft^2 is usually adequate on sites receiving greater than 12 inches of precipitation. Restoration may be slow especially if undesirable plants such as cheatgrass dominate the understory. If the site has fewer than the required minimum of perennial understory plants the community should be seeded to a desired mix of species adapted to the site. Broadcasting seed directly on the bare soil surface will have little success, however, spreading seed in slash will greatly increase the success of seedling establishment.

The decision to burn or when to burn sites following cutting should be determined by the objectives. Generally the lower the site condition the more important the juniper slash. Burning to reduce new trees establishing on the site probably can be delayed until the trees are approaching a height of 10 ft. However, the response of the understory vegetation and objectives should dictate when and if fire should be used on a site that has been thinned.

Aspen Communities

In the past 100 years in the Pacific Northwest, aspen has steadily declined due to conifer encroachment. In the semi-arid shrub region, western juniper is rapidly invading aspen groves below 7,000 ft. The conversion of aspen groves to western juniper is primarily attributed to the reduced role of fire and to overgrazing by wild and/or domestic large herbivores. Both fire and cutting are effective tools for restoring aspen stands. However, if overgrazing by large herbivores remains a problem the stands will remain at risk. Aspen communities provide considerably greater foraging and nesting opportunities for all types of wildlife than former aspen groves that have become converted to dense juniper. Aspen stands have greater diversity and abundance of insects, and provide a higher quality forage than juniper woodlands. A wildlife species highly dependent on aspen is the red-naped sapsucker.

Other Considerations

On sites where the understory is in very poor condition the question should be asked if the poor health of the understory is due to the increase in juniper dominance, past and or current management practices, or a combination of both. If current practices such as grazing are a problem, thinning juniper trees will not restore the site. Grazing management should be based on maintaining a healthy stand of plants and a healthy functioning watershed. Grazing management should focus on minimizing soil loss and enhancing infiltration rates. It should also focus on allowing plants to capture sufficient water, nutrients, and sunlight, and retain sufficient plant cover and litter to protect both plants and soils. Sites in poor condition that have been thinned should not be grazed, and managed as a new seeding. The number of years of deferment will depend on the condition of the site prior to treatment and weather conditions in subsequent years following treatment. The response of the plant community in the years following treatment will provide the best indicator as to when to begin grazing.

Water

Water is frequently limiting for wildlife in many juniper communities. Making water available can greatly increase wildlife use of open juniper communities. This is particularly true in the pumice region where surface water is scarce. Guzzlers are an effective tool for increasing the availability of water for wildlife.

CONCLUSIONS

Maintaining a balance of juniper trees with other plant forms such as shrubs, grasses, and wildflowers will provide the greatest opportunity for the maximum number of wildlife species at the community level. At the landscape level the greatest diversity of wildlife species will occur where there is a mosaic of communities in various stages of succession. Shrub communities should be managed for various successional stages from early grassland communities, which follow a fire, to shrub grassland to open mid-successional juniper communities. However, once the woodlands have succeeded to the late or closed stages of development, plant diversity, berry production, and the opportunity to burn have greatly decreased. The expense of cutting also increases with increasing tree density and tree size. In some cases a limited amount of dense closed woodlands forming corridors or small islands may be the desired objective. To minimize the risk of soil loss, management of these dense stands should be on deep well drained soils where an adequate herb layer can persist. However, the majority of these shrub steppe landscapes, which were open prior to the time of settlement, should be managed in a more open state.

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Table 2. Checklist: optimizing wildlife habitat in juniper woodlands.

Site

a. Identify the stage of woodland development (see Table 1, pg. 14).

b. Identify the plant community and condition.

Thinning to maintain mid successional stage woodland development:

Benefits – retains understory shrubs and herbs and potential berry crops; increases growth of desirable shrubs such as bitterbrush and mountain mahogany; maintains the health of old trees; increases ground cover reducing overland flow and increasing infiltration rates.

a. Retain old growth juniper trees.

b. Thin to approximately 5 to 12 full size trees per acre or less than10% tree canopy cover; 5% or less tree cover on less productive sites. On poor condition sites – cut and spread limbs across the site to increase ground cover to enhance site restoration.

c. Broadcast seed beneath slash on sites with less than two desirable perennial plants/10 ft^2 (1/10 ft^2 where annual precipitation is greater than 12 inches).

Burning

a. Burning should not be considered (and is probably not practical) on sites lacking desirable understory shrubs and herbs and dominated by introduced annuals such as cheatgrass and medusa-head.

b. Burning should be considered where sites are in good condition, adequate fuel is available, the threat of exotic annuals is low, and large land areas need to be treated.

IMPORTANCE OF WESTERN JUNIPER COMMUNITIES TO SMALL MAMMALS[1]

Mitchell J. Willis and Richard F. Miller

SUMMARY

We investigated the composition and relative abundance of small mammals in western juniper (*Juniperus occidentalis*) woodlands of southeast Oregon in the spring of 1993 and 1995-97 by snap trapping recently cut sites, shrub dominated sites, and adjacent uncut (both mid-successional and old-growth) juniper sites. The number of captures was almost always higher in the cut sites than in the uncut sites, but results were mixed in the shrub/tree comparisons. The number of species captured was higher in shrub sites over old growth woodland sites. We believe structure provided by robust understory vegetation and the overhanging juniper skeletons provided superior security and forage for small mammals in the cut and dropped sites.

INTRODUCTION

The issue of juniper encroachment, conversion, and subsequent impacts of community structural changes on small mammals has been of increasing interest by resource managers in recent years. Although research has been conducted on the direct effects of juniper on forage productivity, plant composition and structure, and impacts on big game, little has been directed toward small mammals associated with western juniper (*Juniperus occidentalis*).

Of the estimated 341 animal species found in southeastern Oregon (Maser et al. 1984), 95 have been reported to occur in juniper steppe (Puchy and Marshall 1993). Juniper steppe is defined as western juniper woodlands, typically having a sagebrush (*Artemisia sp.*) understory. The western juniper/sagebrush/bunchgrass community had the third largest number of the 341 total wildlife species from the 16 general plant communities Maser et al. (1984) describe in southeastern Oregon. Puchy and Marshall (1993) also report large numbers of wildlife use juniper steppe. However, both of these reports were based on minimal data and were written as guidelines. These reports lumped a broad range of transitional phases of juniper succession in shrub-steppe across a wide variety of environmental variables, all of which affect plant community structure, composition, and function (Miller et al., 1998). Juniper-shrub steppe communities described in the literature are typically shrub-steppe communities in various stages of woodland conversion. Shrubs and some perennial grasses and forbs are lost as woodlands approach full development, changing the structural characteristics of the understory (Miller et al. 1998).

Few studies have evaluated the effects of juniper woodland development or conversion in shrub steppe communities on nongame use, and most of these have been conducted in pinyon (*Pinus sp.*) juniper communities. Baker and Frischknecht (1973) examined small mammal changes relative to clearing and seeding in pinyon-juniper communities in Utah, they report large increases in white footed-deer mice (*Peromyscus maniculatus*) and Great Basin pocket mice

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[1] In: Monsen, S. B., R. Stevens, R.J. Tausch, R. Miller, S. Goodrich. 1998. Proceedings: Ecology and management of pinyon-juniper communities within the interior west. 1997 Sept. 15-18 Provo, UT. Gen. Tech. Rep. INT-GTR-000. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station.

(*Perognathus parvus*) in cut areas for the first three years after treatment, followed by a reduction to a population level still above that before treatment. Turkowski and Reynolds (1970) report 1.2-4.0 times as many rodents on treated (cut) plots over untreated plots three years after treatment in the same type on the Kaibab Plateau in Arizona. In pinyon-juniper woodlands of northeast Nevada, Mason (1981) report rodent numbers increased while species diversity decreased on burned pinyon-juniper sites during the first two years following treatment; both bird numbers and diversity increased on burned areas in these woodlands over comparable unburned areas. O'Meara et al. (1981) noted that small mammal abundance in Colorado was higher in chained pinyon-juniper woodland than in control plots. They suggested adverse effects on nongame wildlife could be minimized by favoring survival of shrubs and young trees, retaining selected cavity trees, and limiting widths of clearings when chaining. O'Meara et al. (1981) also reported higher bird densities in unchained areas than in chained areas. Sedgwick and Ryder (1987) report that richness of small mammal species and total captures were greater on chained versus unchained plots of pinyon-juniper in Colorado, and seven of the 16 most common bird species in the area used the control plot more, while only one used the chained plot more. Severson (1986) reports total numbers of small mammal species significantly greater on all treated areas compared to untreated pinyon-juniper woodlands 13-18 years post-treatment. Individual species and groups respond differently to the tree removal manipulations and methods of slash disposal. Grassland rodents as a group are more abundant where the overstory and slash had been removed; however, wood rats (*Neotoma sp.*) and brush mice (*Peromyscus boylii*) are greatest where the slash remained on the site. Pinyon (*P. trueii*) and rock mice (*P. difficilus*) prefer the thinned site, where slash remained on site. Austin and Urness (1976) report few differences with respect to total rodent numbers and weight in a comparison among seven pinyon-juniper types.

Probably the apparent conflicting results of small mammal and bird responses to woodland treatment is probably largely due to the limited vegetation data collected in these studies. Juniper and pinyon-juniper woodlands occur across a wide variety of spatial and temporal conditions in the Intermountain west (Miller et al. 1998, Tausch et al. 1998). Woodland structure and composition before treatment and succession after treatment will likely affect small mammal and avian populations.

We investigated small mammal and bird composition and relative abundance in southeast and central Oregon, northwest Nevada, and northeast California in 1993 and 1995-97. This report compares small mammal populations between cut and uncut stands of mid-aged western juniper woodlands, old growth woodlands with adjacent shrubland, and also mid-aged stands with the old growth stands in southeast and central Oregon.

STUDY AREAS

Page Ranch: Closed Woodland vs. Cut

The Page Ranch study area was located in Grant County, Oregon, along Warren Creek at about 4,600 ft with northwest 20 percent slopes. Treated sites were about 25 acres in size. Vegetation was a community of mountain big sagebrush *(Artemisia tridentata ssp. vaseyana)*/ Idaho fescue (*Festuca idahoensis*). The juniper stand was fully developed, about 40 percent canopy cover, 100-130 trees/acre, with sparse understory shrubs. Perennial grasses and forbs had higher cover values in cut sites but were common on both. Three treatment blocks were cut and the trees left in place during the fall of 1992. Sampling commenced in the spring of 1993 and was repeated in 1995-97.

Krumbo Ridge: Mid-transitional Juniper-Shrub Steppe vs. Thinned

The Krumbo Ridge study area was located at about 5,000 ft on Steens Mountain, Harney County, Oregon. Slopes were generally northerly and less than 2 percent. The three uncut sites were mid-transitional woodlands with 10-20 percent tree canopy cover and 100 trees/acre, 7-15 percent shrub cover, and 5-10 percent perennial herbaceous cover. The three cut sites were thinned to 2-3 trees/acre (1-2 percent canopy cover) in the spring of 1995. Understory vegetation was characterized by mountain big sagebrush and Idaho fescue.

Juniper Mountain: Old Growth vs. Shrub Steppe

The Juniper Mountain study area was located in Harney and Lake Counties at about 6,000 ft. All sites were on southeast aspects with 20 percent slopes. The woodland sites were old growth ranging from 400 to 1,000 years old and had 30-35 percent crown closure. Dead and down material was relatively abundant with many cavities. Shrub cover under the trees was less than 1 percent. The shrub sites had 35 percent cover of mountain big sagebrush. Abundant herbaceous plants were bluebunch wheatgrass (*Agropyron spicatum*), Thurber's needlegrass (*Stipa thurberiana*), and bottlebrush squirreltail (*Sitanian hystrix*).

Green Mountain: Old Growth Juniper Woodland vs. Shrub Steppe

The Green Mountain study area was located in Lake County Oregon at 5,000 ft. Sites were southeast aspects with <5 percent slopes. The woodland canopy was more open with slightly less dead and down woody material and fewer cavities than Juniper Mountain. Tree canopy ranged from 10-15 percent, shrub cover was <1percent, and herbaceous cover 10 percent. The adjacent shrub sites burned about 50 years ago. Shrub cover was 15 percent and herbaceous cover was 10-15 percent.

METHODS

Small-mammal trapping was conducted in permanently marked grids centered within each site. Museum Special traps were set within 1m of flags placed in a 10 x 10 array at 5 m intervals totaling 100 traps/grid. Traps were baited with peanut butter and rolled oats. At each study area, treatments and controls were repeated three times resulting in 3,000 trap-nights for each study area. Traps were set on day 1, checked in the early morning on days 2-4, and pulled after checking on day 5. Trapping was conducted May-early July. The status of each trap was recorded daily. Captured mammals other than white-footed deer mice were placed in plastic bags with plot number, treatment, station, and date recorded on the outside, cooled on ice, and later identified. White-footed deer mice were removed from traps and left in the area in deference to hantavirus concerns.

Museum Special snap traps were used in lieu of livetraps. While capture selectivity may exist among trap types (Fowle and Edwards 1954), snap traps have been utilized extensively and effectively (Johnson and Keller 1983). Snap traps were found more effective than pitfall traps for deer mice and chipmunks (*Tamias sp.*) (McComb et al. 1991), both common to juniper woodlands (Johnson and Keller 1983). Hayward and Hayward (1995) found capture rates between pit and snap traps are generally quite similar for the most common species in their work in central Idaho. We would liked to have used pitfall and rat-traps as well, but logistics and finances limited effort to Museum Specials. Significant as used in the text is $p \geq 0.05$.

RESULTS

Fourteen species representing four orders of small mammals were captured (Table 1) from the 30,000 trap-nights of study. We had 898 total captures ($\bar{x}$ = 34 trap-nights/capture, range= 16-86). The white-footed deer mouse was the most often captured species (n̲ = 614; 68.4 percent), followed by yellow pine chipmunk (n̲ = 122; 13.6 percent, and Great Basin pocket mouse (n̲ = 81; 9 percent, Table 2).

Cut vs. Uncut

Seven trapping sessions were conducted comparing cut vs. uncut sites (Page Ranch and Krumbo Ridge). Although the difference was not significant, between cut and uncut sites across the two study locations, in all but one instance (Krumbo Ridge 1995), there were more captures in the cut blocks (uncut $\bar{x}$ = 31.71, cut $\bar{x}$ = 57.43). The number of species encountered (species richness) in trapping sessions was three times greater in cut sites, two times in uncut sites, and tied twice. The greatest number of species encountered (uncut treatment at Page Ranch, 1995) was eight, and two was the lowest (2 other uncut treatments at Page Ranch, 1993,1997). Mountain pocket gophers (*Thomomys talpoides*) and western jumping mice (*Zapus princeps*) were not caught in cut sites.

The lack of response of small mammals to cutting across the two areas is partially due to differences in woodland structures between the two locations. When evaluated within location, differences in small mammal abundance and diversity show up where the juniper woodland is fully developed and shrubs have been lost in the understory.

Eight different species were captured over the four years of study in the closed post-settlement juniper woodland and adjacent cut plots at Page Ranch. Montane meadow mice, white-footed deer mice, and yellow pine chipmunks were significantly more abundant in cut sites in 1995. This pattern held in 1996 as well. There were consistently (but non-significant) more captures in the cut blocks (uncut $\bar{x}$ = 28.5, cut $\bar{x}$ = 71.5) over the four years of study. However, in the mid-transitional juniper-shrub steppe and thinned sites at Krumbo Ridge, no distinctions or consistent patterns were noted between treatments or years. Shrub-steppe structural characteristics were present in both treatments. Nine different species were captured over the three years of study.

Table 1. Small mammals captured at Page Ranch, Krumbo Ridge, Juniper Mountain, and Green Mountain in Eastern Oregon, 1993, 1995-97. Small mammal densities by species followed by a * were significantly different ($p = 0.05$) between treatments within years.

Page Ranch (Uncut vs. cut)	1993		1995		1996		1997	
Long-tailed meadow mouse	0	0	1	0	0	0	0	0
Montane meadow mouse	0	1	2	34*	0	5*	0	0
Bushy-tailed wood rat	0	0	1	0	0	0	0	0
Canyon mouse	0	0	1	1	0	0	0	0
White-footed deer mouse	19	23	36	86*	35	72*	9	18
Great Basin pocket mouse	1	1	2	5	1	4	0	3
Yellow pine chipmunk	0	4	1	15*	1	11*	2	3
Northern pocket gopher	0	0	1	0	1	0	0	0
Western jumping mouse	0	0	0	0	0	0	0	0
Krumbo Ridge (Uncut vs. Cut)								
Long-tailed meadow mouse			0	0	0	3	0	2
Montane meadow mouse			1	1	1	1	0	1
White-footed deer mouse			15	12	25	35	25	27
Great Basin pocket mouse			2	1	8	6	6	3
Vagrant shrew			0	0	0	1	0	0
Yellow pine chipmunk			5	8	13	8	3	4
Mountain cottontail			0	0	0	0	0	3
Northern pocket gopher			1	0	2	0	0	0
Western jumping mouse			0	0	1	0	0	0
Juniper Mountain (Shrub vs. Tree)								
Long-tailed meadow mouse					0	0	3	0
Montane meadow mouse					2	0	5	0*
Ermine					0	0	1	0
White-footed deer mouse					59	47	26	11
Great Basin pocket mouse					9	2*	12	1*
Yellow pine chipmunk					3	4	1	16*
Northern pocket gopher					1	0	0	1
Green Mountain (Shrub vs. Tree)								
Ord kangaroo rat							1	0
Desert wood rat							0	1
White-footed deer mouse							10	24*
Great Basin pocket mouse							12	2*
Yellow pine chipmunk							1	19*

Shrub vs. Woodland

Three trapping sessions were conducted comparing shrub-dominated sites with adjacent old growth juniper woodlands. There were significantly more mammals captured in shrub sites at Juniper Mountain (shrub $\overline{x} = 61$ captures, woodland $\overline{x} = 41$ captures), while at Green Mountain, more were taken in woodland sites (shrub = 61 captures, woodland = 41 captures). Eight species were captured in shrub sites, and five in woodland sites.

At the Juniper Mountain area, there were significantly more Great Basin pocket mice in the shrub sites in 1996. In 1997, there were significantly more montane meadow mice, Great

Basin pocket mice, and fewer yellow pine chipmunks in the shrub sites. Although not significant, there were more white-footed deer mice in the shrub sites both years (59 vs. 47 captures in 1996, and 26 vs. 11 in 1997).

At Green Mountain, significantly fewer white footed deer mice and yellow pine chipmunks, and significantly more Great Basin pocket mice were found in the shrub sites.

Post-settlement vs. Old Growth Juniper Woodland

Five different species were captured in old growth sites and three in mid-successional sites in 1997. At old growth sites, there were significantly more yellow pine chipmunks than in post-settlement woodlands in 1997 (n = 5 mid, 35 old growth). There were generally more Great Basin pocket mice in the mid-successional woodland sites which contained a shrub understory (n = 6) than in the old growth (n = 3). White-footed deer mice were about equal between the two types (n= 34 and 35 for mid vs. old respectively).

CONCLUSIONS

Our capture rates were highly variable among sites within treatments, among years, and among areas. This undoubtedly caused the lack of significance among many comparisons. With four years of sampling at Page Ranch, we hoped to find trends in composition and abundance of small mammals post-cutting. We expected some sampling "noise" but not of the magnitude encountered. Sedgwick and Ryder (1986) encountered 11-fold changes in capture rates among years in their pinyon-juniper sampling, and cited several others who had documented similar results. We hope to periodically sample at least the Page Ranch site to search for longer term trends, as those found by Severson (1986) who reported total numbers higher in manipulated sites, but a variety of individual species responses in New Mexico 18 years post-treatment. Baker and Frischknect (1973) snap-trapped a chained and seeded pinyon-juniper range, then concluded that deer mice and pocket mice populations increased through the second year post-treatment, and then dropped to levels still above uncontrolled. This pattern may have occurred at Page Ranch, but we couldn't separate population patterns from noise. O'Meara et al. (1981) found higher small mammal abundance but fewer species in 1-, 8-, and 15-year old chained sites over controls. They also pointed out that leaving blocks of unchained vegetation within pinyon-juniper control areas should maintain woodland dependent species while providing increased total numbers of small mammals in treated areas.

The total captures and the number of species captured in our study were higher in the cut sites than in the uncut sites, comparable to the findings of Sedgwick and Ryder (1986) and Severson (1986), although their work was in pinyon-juniper, and the treatments were chainings.

Although we have no data on optimal size of treatment area, our findings concur with others that small openings in pinyon-juniper woodlands (Albert et al. 1994), and in aspen (*Populus tremuloides*) (Christian et al. 1996), can benefit a variety of wildlife. Smallwood (1994) expressed concern that habitat fragmentation might increase site invasibility by exotic birds and mammals through decreased indigenous species richness and abundance. In the case of fragmenting young and mid-aged western juniper woodlands, our work with small mammals suggests that potential site invasibility by exotics may actually be diminished because of increased abundance and richness of indigenous species.

We believe the cut sites, particularly at Page Ranch, had preferred structure to the uncut sites which was provided by increased vigor (cover and height) of herbaceous species, increased seed production in the cut sites (Bates et al. in press), greater species richness, and juniper slash which has persisted five years without noticeable change in size and shape. We propose these sites generally provide increased security and forage for small mammals. The lack of differences at Krumbo Ridge was probably due to understory structure being similar between the two treatments. Woodland conversion had not progressed sufficiently to exclude shrubs. Old growth sites typically had a greater variety of species than young juniper woodlands. This may be attributable to the more structurally complex vegetation compared to closed post-settlement woodlands (Miller et al. 1998).

In our opinion, opening stands of western juniper in southeast Oregon by cutting down and leaving trees or thinning does not substantially affect the small mammal component in the area. The Great Basin pocket mouse appears to be the most sensitive species to the loss of shrubs during the latter stages of concern from shrub steppe to juniper woodland. However, other species such as wood rats are favored by the presence of juniper trees in the stand. For the maintenance of maximum structural diversity in post-settlement stands, shrub steppe communities should be managed through early- to mid-woodland succession (Miller et al. 1998). If conversion crosses a threshold, moving into late and closed woodlands, structural complexity and plant species diversity in the understory decline, resulting in shifts in small mammal population dynamics.

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AFFECTS OF COMMUNITY STURCTURE ON AVIAN POPULAITONS IN JUNIPER WOODLANDS AND SHRUB STEPPE: WORK IN PROGRESS

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SUMMARY

A large proportion of the juniper/sagebrush/bunchgrass communities are in transition from sagebrush/bunchgrass to closed juniper woodlands. Closed woodlands typically have few to no shrubs in the understory and may or may not have an intact herbaceous understory. However, the majority of avian surveys and studies in juniper communities have treated the woodlands generically despite the large degree of variability in community structure and composition. The greatest abundance and diversity of birds is usually found in the mid successional woodlands or open old growth woodlands with an intact shrub steppe component in the understory. As the woodland closes and shrubs decline ground and shrub nesters vacate the site.

INTRODUCTION

In southeastern Oregon, the greatest number of wildlife species have been reported in sagebrush/bunchgrass and juniper/sagebrush/bunchgrass communities, with the exception of riparian areas (Maser et al. 1984). A large proportion of the juniper/sagebrush/bunchgrass communities are in transition from sagebrush/bunchgrass to closed juniper woodlands. Closed woodlands typically have few to no shrubs in the understory and may or may not have an intact herbaceous understory. The juniper/shrub/bunchgrass communities with the high wildlife diversity described in Maser's et al. (1984) report are early to mid successional communities proceeding to closed woodlands. In addition to changes over time due to woodland succession, environmental factors such as soils, topographic position, elevation, and disturbance history also influence woodland structure and composition. Changes in woodland structure due to succession or site characteristics affects wildlife habitat for a broad array of species. For example, open juniper woodlands in early to mid succession with an intact shrub bunchgrass understory typically have the greatest structural diversity with multiple plant layers (e.g. trees, shrubs and grasses) compared to a shrub grassland or closed juniper woodland. Community structure, particularly vegetation layers, have been closely related to avian diversity. However, the majority of avian surveys and studies in juniper communities have treated the woodlands generically despite the large degree of variability in community structure and composition. We must consider the dynamics of juniper woodlands to effectively manage for avian populations.

Several studies evaluating avian populations in semi-arid woodlands have been conducted in the pinyon-juniper zone with mixed results. Mason (1981) reported bird numbers and diversity increased on burned areas over comparable unburned areas in pinyon-juniper woodlands of northeast Nevada. O'Meara et al. (1981) found bird densities to be higher on unchained than chained areas in pinyon-juniper woodlands.

They believed the adverse effects of chaining on non-game wildlife could be minimized by favoring survival of shrubs and young trees, retaining selected cavity trees, and limiting widths of clearings when chaining. Sedgwick and Ryder (1987) found 7 of the 16 most common bird species in a pinyon-juniper study in Colorado used the untreated woodland plot more, while only 1 used the chained plot more. However, these studies failed to report the structure and composition of treated and untreated woodlands.

In a recent symposium on the management of pinyon-juniper in the Intermountain West, two studies evaluating avian diversity and abundance in pinyon juniper woodlands conducted across Utah, failed to identify stand structure and age. In one of the studies, two of the 17 stands contained several shrub and ground nesting birds including Brewer's and vesper sparrows. Both stands were mid succession woodlands with a shrub layer compared to the remaining closed stands where shrubs were absent. A third stand contained a high percentage of cavity nesters, attributed to the unusually large size of the juniper trees. However, the stand was an old growth woodland containing considerable dead wood and cavities. A second study compared avian diversity and abundance among different conifer types including a pinyon-juniper woodland. Among the seven upland forest cover types sampled, the pinyon-juniper woodland ranked second in bird abundance, third in species richness, and third in overall diversity. The woodland type ranked first in percentage of obligate and semi-obligate species. Although not identified in the study, the pinyon-juniper woodland was an old growth stand with a diverse understory.

We are currently conducting avian surveys across central and southeastern Oregon, northwestern Nevada, and northeastern California (Fig. 1). Our overall goal is to evaluate the affects of succession in shrub steppe juniper woodland communities on bird density, diversity, species richness, and composition during the breeding season and during the winter. We also intend to relate the relative abundance of birds with structural components of the vegetation. Communities being evaluated include grasslands (recently burned sagebrush bunchgrass), sagebrush grassland, mid successional sagebrush bunchgrass woodlands, closed postsettlement woodlands, and old growth juniper woodlands. In addition to burning, we are also evaluating the affects of juniper thinning and clearcut with slash left in place on avian populations. We currently have from one to three years of data across 10 locations (Fig. 1). We are currently in the process of analyzing the data collected during the past three years and will continue to conduct bird surveys in 1999 and 2000.

STUDY AREAS

Bird surveys have been conducted in 10 areas: Page Ranch near John Day, Krumbo Ridge and Deep Creek on Steens Mountain, Juniper Mountain, Green Mountain, Pine Mountain, Badlands, and West Butte in Oregon, Devil's Garden in Modoc California and Sheldon National Wildlife Refuge in Nevada (Fig. 1).

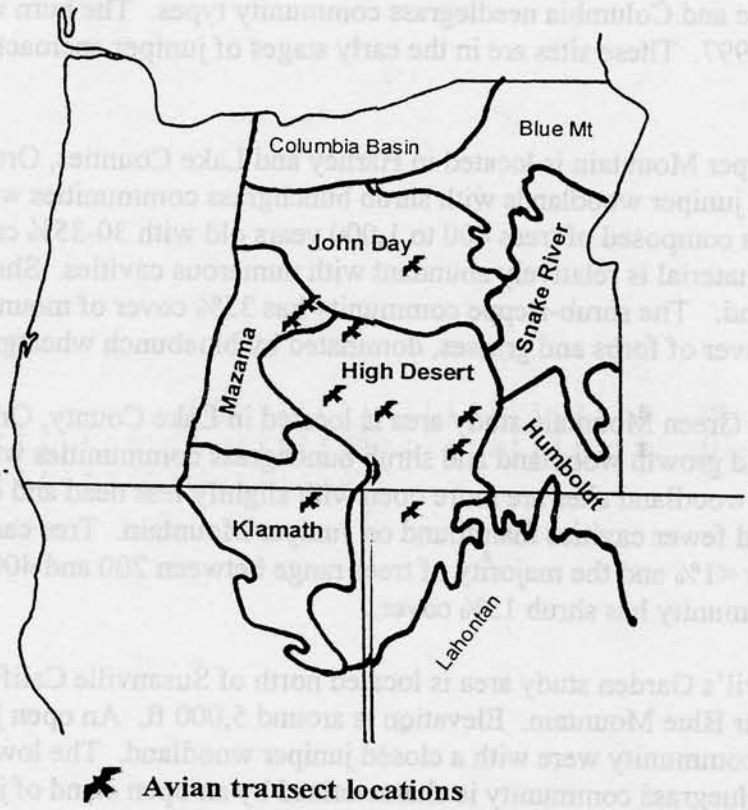

Figure 1. Avian bird study areas located across four ecological provinces. Ecological provinces are derived from Anderson 1956, Cronquist et. al. 1972, and Bailey 1994.

The Page Ranch is located along Warren Creek approximately 8 miles northwest of Mount Vernon in Grant County, Oregon at 4,600 ft. The area is occupied by a late successional juniper woodland with about 40% tree canopy cover and 100-130 trees/acre. Understory vegetation is primarily a mix of grasses and forbs with few shrubs. Three 50 acre plots were marked out on a northwest slope. Half of each plot (approx. 25 ac) were cut to compare both plant and avian response to tree removal. Trees were cut and left in place.

Krumbo Ridge is located on the northern end of Steens Mountain at about 5,000 ft. in Harney County, Oregon. Treatments are thinned and uncut woodlands. Uncut sites are in mid to late woodland succession with 10-20% tree cover and 100 trees/acre. Shrub cover varied between 7-15%, although well over half of the shrub canopy, both sagebrush and bitterbrush, were dead. Herbaceous cover was 5-10%. The three cut sites were thinned to 2-3 trees/acre (1-2% canopy cover) in the spring of 1995.

The Deep Creek area is located on the south end of Steens Mountain near the headwaters of the Blitzen in Harney County, Oregon, at an elevation of 6200 ft. Comparisons are being made between burned and unburned mountain big sagebrush and

Idaho fescue and Columbia needlegrass community types. The burn was conducted in the fall of 1997. These sites are in the early stages of juniper encroachment with < 1% tree cover.

Juniper Mountain is located in Harney and Lake Counties, Oregon at 6,000 ft. Old growth juniper woodlands with shrub bunchgrass communities were compared. The woodland is composed of trees 400 to 1,000 years old with 30-35% crown closure. Dead and down material is relatively abundant with numerous cavities. Shrub cover is <1% in the woodland. The shrub-steppe community has 35% cover of mountain big sagebrush and 12% cover of forbs and grasses, dominated by bluebunch wheatgrass.

The Green Mountain study area is located in Lake County, Oregon at 5,000 ft. Adjacent old growth woodland and shrub bunchgrass communities were compared. The old growth woodland sites are more open with slightly less dead and down woody material and fewer cavities than found on Juniper Mountain. Tree canopy is 10-15% and shrub cover <1% and the majority of trees range between 200 and 400 years. The shrub steppe community has shrub 15% cover.

Devil's Garden study area is located north of Susanville California on the USFS 47 road near Blue Mountain. Elevation is around 5,000 ft. An open juniper low sagebrush community were with a closed juniper woodland. The low sagebrush Sandberg bluegrass community is characterized by an open stand of juniper, <5% canopy cover.

The Sheldon Refuge area is located in northern Washoe County, Nevada at 6,400 ft. The community type being evaluated is a mountain big sagebrush/Columbia needlegrass community type. Three unburned sites and three burned sites were evaluated. The burn communities were in the third growing season following fire.

In Central Oregon (Horse Ridge just east of Bend and Fort Rock), a two year intensive bird study is being conducted comparing avian populations among recently burn shrub bunchgrass, shrub bunchgrass, mid-succession juniper woodland, and old growth woodlands. Bird surveys are being conducted during the winter and spring. Bird transects are distributed from the Badlands just north of highway 20 east of Bend, south to near Christmas Valley. First year has been completed.

METHODS

Bird inventories are being conducted on each study area using the variable circular-plot technique outlined by Reynolds et al. (1980). Surveys commence within 0.5 h of sunrise and do not extend beyond 4 h after sunrise. At each point, all birds seen or heard as well as cluster size, and distance from station center are recorded for a 5 minute period following a 1 minute period for acclimation (Verner 1988). On mornings with winds > 10 mph or precipitation, surveys are not conducted. Plots were surveyed 4 times during a one week period with the exception of the intensive bird survey where plots were survey 5 times during May and June, and monthly December through April.

The Shannon-Weiner diversity index was calculated from field observations for each plot summing across days.

DISCUSSION

The following are a few general conclusions based on data summaries. Data have not yet been statistically analyzed. In cut versus uncut woodlands avian diversity and density were higher in cut juniper treatments than in uncut blocks at both Page Ranch and Krumbo Ridge study areas. We suspect that cutting juniper and leaving slash within the stands at the Page Ranch created additional structural diversity for the birds. Cover of herbaceous vegetation greatly increased on the cut plots. On the Krumbo Ridge site, tree selective species, including the Dark-eye junco, mountain chickadee, chipping sparrows, and flycatchers were more abundant on uncut plots but also used the thinned stands. Shrub and ground nesting birds including Green-tailed and Rufous-sided Towhees were more abundant on the thinned plots. Chipping sparrows were probably the most ubiquitous species across treatments. They appeared to be highly dependent on the slash in the cut sites. We also noted the majority of birds in the uncut closed woodland on the Page Ranch used the edge adjacent to the cut plots. In the cut plots birds were evenly distributed across the treatment.

Closed old growth woodlands were compared to shrub steppe communities at Juniper Mountain, Green Mountain, and Devils Garden. Avian diversity among these juniper stands and shrub steppe communities were notably different only at Juniper Mountain. At Green Mountain and Devil's Garden the indices were all very close although species composition was different between woodlands and shrub bunchgrass communities. Chipping sparrow density was consistently higher in the woodland treatments. Western Meadowlarks and Green Tailed Towhees were found exclusively in the shrub treatments in all 3 areas.

In the Central Oregon transects (Bend and Fort Rock areas), the highest bird numbers, approximately 45, and diversity indices occurred in open old growth woodland and early successional postsettlement woodland, compared to all of the other transects we have surveyed. Both communities contained less than 15% tree cover, and a shrub grassland understory is intact. This compares to 16 to 26 bird species in closed juniper woodlands with no shrub layer.

In recently burned and shrub steppe communities on the Sheldon National Wildlife Refuge we recorded approximately 21 and 24 bird species, respectively. However, the burned plots contained a greater density of birds (169/ha) compared to unburned (107/ha). Diversity indices were similar between treatments but composition was different.

A preliminary summary of data shows avian diversity among the different community types was greatest in mid succession shrub steppe woodland communities and open old growth stands, where the tree canopy is open and the shrub steppe understory is still intact. The composition and structure of the understory appears to be a key factor influencing avian abundance and diversity in these woodland communities. However, if juniper woodlands are allowed to fully develop, both the shrub layer and potential berry crops will be lost as well as the opportunity to burn these stands. Maintaining a balance of trees with other plant forms such as shrubs, grasses, and forbs will provide the greatest opportunity for the maximum number of avian species at the community level. In addition, the maintenance of different successional stages in shrub steppe communities, from recently burned to fully developed sagebrush bunchgrass stands will increase opportunities for ground and shrub nesting birds, and foraging for a broad array of bird species.

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